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Treatment of Lake Charles East, Indiana Sediments with Fly Ash

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16. ABSTRACT

This report contains information relating to the degree of effectiveness of the treatment of eutrophic lake sediments with a specific power plant fly ash. The treatment was preceded by the diversion of the major nutrient sources outside of the drainage basin. Data on both chemical and biological changes are documented.

The study area was Lake Charles East, an 8.7 ha lake in northeastern Indiana. Treatment of approximately one-third of the sediments with fly ash and lime took place during the summer of 1975. Follow-up studies indicated reduced release of phosphorus during peak summer release periods for treated sediments. Mass balance modeling indicated a net reduction in long term phosphorus levels of 20% over levels without sediment treatment. If all sediments had been treated, the steady state phosphorus levels were predicted to decline by 61% over non-treatment levels. The phytoplankton community composition changed from one dominated by blue green species virtually year round to one in which the more classical successional pattern of diatoms-greens-blue greens took place. Cryptophytes became much more important in the post-treatment period. Zooplankton communities showed only short term effects from the treatment. Benthic organisms, dominated by midge larvae, were not affected.

Total heavy metal concentrations increased slightly in the treated sediments, however, soluble levels in both the water column and the sediment interstices were not elevated.

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FOREWORD

Effective regulatory enforcement actions by the Environmental Protection Agency would be virtually impossible without sound scientific data on pollutants and their impact on environmental stability and human health. Responsibility for building this data base has been assigned to EPA's Office of Research and Development and its 15 major field installations, one of which is the Corvallis Environmental Research Laboratory (CERL).

The primary mission of the Corvallis Laboratory is research on the effect of environmental pollutants on terrestrial, freshwater, and marine ecosystems; the behavior, effects and control of pollutants in lake systems; and the development of predictive models on the movement of pollutants in the biosphere.

This report describes the results of an evaluation on the effectiveness of using fly ash in restoring eutrophic lakes by inactivating nutrients and preventing cycling of phosphorus from the sediments.

James C. McCarty
Acting Director, CERL

ABSTRACT

This report contains information relating to the degree of effectivenss of the treatment of eutrophic lake sediments with a specific power plant fly ash. The treatment was preceded by the diversion of the major nutrient sources outside of the drainage basin. Data on both chemical and biological changes are documented.

The study area was Lake Charles East, an 8.7 ha lake in northeastern Indiana. Intensive monitoring was begun in mid-1974. Phosphorus inputs were reduced by 90% shortly after this time. Treatment of approximately one-third of the sediments with fly ash and lime took place during the Summer of 1975. Follow up studies indicated reduced release of phosphorus during peak summer release periods for treated sediments, although a small autochthonous layer, presumably due to post-treatment biological activity in the water column, developed above the fly ash. Mass balance modeling indicated a net reduction in long term phosphorus levels of 20% over levels without sediment treatment. If all sediments had been treated, the steady state phosphorus levels were predicted to decline by 61% over non-treatment levels. The dominant mechanisms associated with the sediment effects were chemical alteration of the sediments and physically increasing the diffusional path for P release.

As phosphorus levels in the water column decreased, the phytoplankton community composition changed from one dominated by blue green species virtually year round to one in which the more classical successional pattern of diatoms-greens-blue greens took place. Cryptophytes became much more important in the post-treatment period. Blue green dominance, at the end of the study, was confined to a late summer period. Zooplankton communities showed only short term effects from the treatment and appeared to be more responsive to the changing phytoplankton composition. Benthic organisms, dominated by midge larvae, were not affected.

Total heavy metal concentrations increased slightly in the treated sediments, however soluble levels in both the water column and the sediment interstices were not elevated. Any heavy metal effects, although potentially of importance in this type of treatment, appeared to be masked by other present and past inputs to the system such as lead from highway exhaust, copper from algal treatments, and arsenic from weed control efforts.

This report was submitted in fulfillment of Grant Number R-801245-04-2 by the Department of Civil Engineering of the University of Notre Dame under the sponsorship of the Environmental Protection Agency. Work was completed as of December 31, 1977.

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SECTION I

CONCLUSIONS

1) The fly ash used to treat the sediments in Lake Charles East brought about statistically significant reductions in phosphorus release during anoxic periods in comparison with control sediments.

2) The sediments of Lake Charles East are capable of contributing large amounts of phosphorus to the lake on a seasonal basis after external pollution abatement. This brings about noticeable short term P fluctuations and has lesser long term effects on phosphorus levels in the lake water.

3) Mass balance modeling on phosphorus in the system indicated a net phosphorus retention time slightly less than the hydraulic retention time after abatement. However, a seasonal analysis on phosphorus inputs and outputs showed a large negative internal P reaction constant during the summer season (indicating internal loading from sediments) which was balanced by a large positive constant during the winter. It appears that the determination of a single value for the phosphorus retention time is of limited use in Lake Charles East. This may also be true for similarly stressed systems.

4) The major mechanisms associated with the reduced P release were chemical alteration of the sediments and physically increasing the diffusional path length. With respect to the former mechanism, the smaller amount of amorphous iron oxides (as determined by citrate-dithionite-bicarbonate extraction) in the fly ash layer was an important factor.

5) The appearance of an autochthonous detrital layer above the fly ash which released small amounts of phosphorus has led to the recommendation that supplemental chemicals be used in conjunction with fly ash treatment to limit primary productivity in the immediate post-treatment period.

6) The combination of abatement and partial sediment treatment brought about reductions in both total and soluble phosphorus in Lake Charles East. Long term steady state levels were predicted to be 0.23 mg-P/ℓ. This is a 20% reduction over the projected levels with abatement but without sediment treatment. In addition, cyclic peaks of phosphorus related to sediment release appeared to be dampened. Studies showed that if all of the sediments had been treated, long term phosphorus levels would be reduced 61% over levels without sediment treatment.

7) As phosphorus concentrations in Lake Charles East decreased, significant changes took place in the composition of the phytoplankton community. Total biomass was also less. Cryptophytes became much more dominant after treatment. The spring and summer successional pattern resembled more closely the classical sequence of diatoms-greens-blue greens with blue green dominance being limited to the late summer period.

8) Increased heavy metal concentrations from the fly ash in the sediments did not appear to be a problem in this study.

SECTION II

RECOMMENDATIONS

1) There is a need for a greater degree of classification of both fly ashes (and other particulate materials) and lakes to determine the compatibility of a given system and a specific treatment material for this restoration technique.

2) If the treatment of sediments to effect water quality improvements in polluted lakes is to become more widespread, a more efficient application methodology should be developed. This should be done in concert with a thorough economic analysis.

3) More research on the bioavailability, fate, and effects of heavy metals and trace organics on fly ash added to lake systems should be performed.

4) There appears to be a need to document more extensively species shifts among phytoplankton at high and fluctuating nutrient levels in lakes. Although it was not specifically studied in this research, there is evidence that trace metals may play an important role in this regard.

SECTION III

SCOPE OF THE PROJECT

Previous investigative efforts associated with the effects of pollution abatement on a lake system have suggested that lake sediments may play an important role in the nutrient budget after elimination of major external nutrient sources[1]. It has been shown, and further investigated in this project, that sediments from highly nutrified systems undergo an annual alternating oxidized and reduced state. Oxidized sediments are known to take up or adsorb nutrients from solution and act as a sink, while reduced sediments release nutrients and act as a source. Phosphorus, often a key nutrient in lake algal dynamics, is a chemically reactive specie in natural waters. When introduced to a lake system in excess there is a strong driving gradient for phosphorus toward the sediments via the mechanisms of adsorption, biological incorporation and chemical precipitation followed by settling. The extent to which this nutrient is released during chemically favorable times would be expected to depend upon the amount deposited, the length of time over which the deposition took place and the chemical form of the phosphorus. The relative contribution of the released phosphorus toward the delay of water quality improvements depends in turn on the lake and basin morphology and flushing rate.

This investigation consisted of a small demonstration project and associated studies to reduce the release of phosphorus from eutrophic lake sediments through the addition of power plant fly ash and to assess the effectivenss of the treatment. Fly ash is a fine-grained, largely inorganic residue remaining after the combustion of coal. In the United States approximately 50 million tons are produced annually. The bulk of this quantity must be considered a waste material. Thus, part of the attractiveness of this method of lake restoration is the disposal of fly ash in a useful fashion.

The demonstration site was Lake Charles East, Indiana, a man-made lake approximately 8.7 hectares in surface area. The lake had received secondary treated wastewater effluent for six years prior to abatement in mid-1974. Treatment of approximately one-third of the sediments took place during the summer of 1975. Approximately 1400 tons of fly ash were used. Follow-up studies were conducted through 1977. An attempt was made to partition the lake so as to provide a control water body; however, the barrier which was erected could not withstand the severe stresses imposed by winter weather.

Complete monitoring of the lake water for biological and chemical parameters was made along with extensive experimentation with both treated and control sediments. Sediment studies consisted of laboratory release/sorption work and a full chemical characterization of the sediment phosphorus. A small number of heavy metal analyses were also made in an attempt to assess the effects of these constituents of fly ash on the sediments.

From the data which were collected, nutrient and hydrologic budgets were obtained. These data were used in modeling the system to determine the impact of sediment treatment and predict the long term phosphorus levels in the lake.

SECTION IV
REVIEW OF THE RELEVANT LITERATURE

PHOSPHORUS IN NATURAL WATERS

The most important form of phosphorus from an eutrophication standpoint is orthophosphate and it will be the chemical interactions of this species that will be examined in some detail in this report. Since phosphorus is often found in natural waters in intimate association with oxidation reduction as well as pH sensitive species it is important to view its features in light of both changing redox potential and pH. The tendency is for increased solubility of phosphorus with decreasing pH and redox potential.

A special case of solid partioning is adsorption in which surface metal ions form metal-ion-phosphate bonds by reaction with solution species. It generally occurs between phosphate ions and the less thermodynamically stable polymorphous iron and aluminum compounds that are usually found in nature in lieu of the more discrete crystalline metal phosphate compounds.

THE CHEMISTRY OF PHOSPHORUS IN LAKE SEDIMENTS

In examining forty-eight surficial Lake Erie sediment samples Williams, et al.[2], found that phosphorus was present in three major forms, (1) phosphorus associated with apatite, (2) non-apatite inorganic phosphorus (NAI-P), and (3) organic phosphorus. The apatite fraction was of natural detrital origin. It existed as particles ranging from fine sand to clay in size but mostly as silt-sized particles and was concentrated in near shore sediments. Both NAI-P and organic-P were concentrated in fine-grained sediments accumulating in offshore depositional areas. NAI-P was associated with amorphous hydrated ferric oxide in the oxidized microzone but was present as vivianite ($Fe_3(PO_4)_2.8H_2O$) and possibly other forms also in the reduced zone. It is noted that amorphous hydrated ferric oxide has an iso-electric point (IEP) of pH = 8.5, at pH values below the IEP, it can remove as much as 5% of its own weight of phosphorus in orthophosphate from solution. It is termed here a "ferric oxide-orthophosphate complex." There is as yet no general agreement as to whether this and similar complexes are essentially homogeneous or whether they consist of an intimate mixture of metal phosphate and metal oxide domains.

Syers, et al.[3], in studying the phosphate chemistry in lake sediments noted that variations in the postulated Fe inorganic-P complex accounted for most of the differences in total Fe and total P between sediments including calcareous ones. Evidence obtained from the exchangeability of inorganic-P sediments indicated that it was highly unlikely that discrete phase Fe and Al phosphates exist in sediments. It was found that while the Madison Lakes were supersaturated with respect to hydroxy apatite rather small amounts of apatite were present in surficial sediments from these lakes. There was little or no evidence that hydroxyapatite played a major role in the chemistry of phosphorus in lake waters or sediments. Application of the solubility product principle was found useful for predicting which phosphorus compounds would be thermodynamically stable or which phosphorus compounds could theoretically form. Examining the relationship between phosphorus and other sediment parameters were closely related to total P, indicating that almost all the variability in total-P content was due to variable amounts of extractable (oxalate or citrate-dithionite-bicarbonate (CDB)) inorganic-P.

PHYSICAL TRANSPORT OF PHOSPHORUS IN LAKE SEDIMENTS

Sridharan and Lee[4] suggested two possible ways in which iron associated phosphorus release could occur in an anoxic system. First, is when ferric iron is reduced to ferrous iron and the phosphorus in association with the former is released to the overlying water. The second, is when phosphorus in association with ferrous iron, such as sorbed phosphorus on ferrous sulfide, is kept in contact with a mild leaching solution under anoxic conditions.

Batch tests run with sediments and overlying water by Fillos and Swanson[5] resulted in steady state concentration in the overlying water of 1.9 to 2.1 mg P/ℓ after 9 days under anaerobic conditions and 0.55 mg/ P/ℓ after 7 days under aerobic conditions. In a continuous flow experiment the release rate drops in a step-wise fashion under anaerobic conditions with time. The release of iron was closely related to that of phosphorus.

DiGiano[6] proposed a model for nutrient-transport in and from the sediments. He claimed that the release of nutrients from bottom deposits to overlying water could be conceived of as a phenomena of diffusion and facilitated transport caused by chemical concentration gradient which occur in three steps: (1) migration from deep layers of the deposit to its surface layer, (2) emergence through the surface to the water immediately above, and (3) dispersion throughout the overlaying water. Porosity was said to control the flux of nutrients per unit area of deposit and tortuousity, which is a measure of the "effective" diffusional path, was said to determine the depth of nutrient penetration. A mass balance was made on a unit volume of sediment and the following equation resulted.

$$N\varepsilon A\Big|_x - N\varepsilon A\Big|_x + \Delta x \pm M = [\varepsilon \frac{\partial c}{\partial t} + (1-\varepsilon) \frac{\partial s}{\partial t}] \, A\Delta x$$

where

N = flux of material, moles/cm^2-sec = $D \frac{\partial c}{\partial x}$ (Ficks 1st law)

ε = porosity

A = cross-sectional area, cm^2

$A\Delta x$ = volume of the segment, cm^3

c = concentration accumulated in the solid phase, moles/cm^3

M = amount formed (+), or amount decomposed (-) due to chemical reactions per unit time, moles/sec

Solutions to this equation for various boundary and initial conditions are made and ways are suggested for obtaining experimental verification of the model and its coefficients. Stumm and Leckie[7] also suggested that the rate of phosphorus release by sediments to the overlying water depends primarily on the rate of transfer (i.e., diffusion) through the interstitial water.

Table 1 summarizes several sediment uptake and release studies that are found in the literature.

LAKE RESTORATION

From a study of 425 problem lakes in the United States, Ketelle and Uttormark[16] found that many could not be restored with cursory protective actions alone. Low flushing rates, sediment releases, phosphorus contributions from the atmosphere, land drainage, and other non-point sources compound the eutrophication problem.

Dunst, _et_ _al_.[17], defined lake restoration as "the manipulation of a lake ecosystem to effect an in-lake improvement in degraded or undesirable conditions." The terms rehabilitation, renovation, renewal, and restoration are often used synonymously. The general approach to lake restoration is to:

1. restrict the input of undesirable materials; and
2. provide in-lake treatment for the removal or inactivation of undesirable materials.

Tenney[18] listed several restoration techniques for water bodies, their effects on water quality, bottom sediments, and aquatic plants, and other advantages and disadvantages of each method. A summary of the techniques follows:

1. chemical control of aquatic plants (biocides);
2. mechanical control of aquatic plants (harvesting);
3. biological control of aquatic plants (herbivores);
4. physical control of aquatic plants (varying water levels);
5. elimination of pollutants in hydrologic inputs (wastewater treatment or diversion; erosion prevention; controlled use of agricultural fertilizers);
6. water replacement by displacement (flushing/dilution);
7. draining and subsequent refilling;

TABLE 1

Some Published Values for Release and Uptake of Phosphorus in Lake Sediments

Release Rate mg-P/m^2/day	Uptake Rate mg-P/m^2/day	Oxic(0) Anoxic(A) Temperature(^{o}C)	Reference
3		(0)	
30		(A)	Fillos and Molof (8)
1.2		(A)	Andersen (9)
1.2		(A) 4^o	
2.2		10^o	
3.5		15^o	
9.4		25^o	Anoub (10)
22-49		8^o	Bengtsson (11)
3		(A)20^o	Fillos and Biswas (12)
26		(A)	
1.2		(0)	Fillos and Swanson (5)
0.8		(A)	Serruya _et_ _al._, (13)
.03-.08		(0)	Bannerman _et_ _al._, (14)
0.27			Stumm and Leckie (7)
17.3 ± 4.6		(A)	
	2.0 ± 1.8	(0)	
12.3 ± 3.6		(A)	
	1.4 ± 0.7	(0)	
1.2 ± 0.2		(A)	
	0.2 ± 0.2	(0)	
0.8 ± 0.2		(A)	
	0.6 + 0.1	(0)	Kamp-Nielsen (15)

8. withdrawal to external treatment facility and return;
9. nutrient inactivation using multivalent metal salts (phosphorus precipitation);
10. addition of particulate materials such as sand, clay, or fly ash (to retard nutrient release from sediments);
11. artificial plastic liner material;
12. dredging (to deepen lake; remove nutrient rich sediments); and
13. destratification (to maintain aerobic conditions).

Dunst, et al.[17], described all of these methods and compiled a list of over 1000 lake rehabilitation experiences throughout the world. Another 1000 or more U.S. lakes will benefit from wastewater treatment.

Lake rehabilitation programs frequently employ combinations or modifications of lake restoration techniques depending on particular lake situations.

THE USE OF FLY ASH FOR LAKE RESTORATION

Fly ash is a particulate waste by-product removed from stack gases during the combustion of coal primarily in electric generating plants. Several authors[19,20] have described fly ash and its uses in construction, manufacturing, and agriculture. Production of fly ash is increasing due to growing energy needs and more stringent air pollution control regulations. Ash disposal or utilization is a growing concern[21,22].

Tenney and Echelberger[23,24] and their co-workers reported that the following properties of fly ash made its addition to small lakes appear feasible:

1. great availability at virtually no cost;
2. low grade adsorbency, due to high specific surface area and residual carbon content, removed soluble organics;
3. water soluble extracts of lime and gypsum precipitated phosphorus;
4. rapid settling, due to particulate nature and high specific gravity, helped remove suspended solids; and
5. particulate and pozzolanic character retarded nutrient release from lake sediments.

Yaksich[25] investigated the use of fly ash, clay, silt, and sand to control phosphorus release from Stone Lake sediments. Fly ash was the most suitable material with respect to water quality, settling, and phosphorus release criteria. Clays did not settle as well as fly ash, while silt sand tended to sink below the surface of flocculative type sediments. Not all fly ashes were equally effective, however. In laboratory jars two types of fly ash prevented phosphorus release under anaerobic conditions for up to five months. One fly ash sank below the sediments and another did not prevent phosphorus release. A layer of fly ash 2 - 5 cm thick was required to prevent disruption by gas bubbles released from the sediments. Higgins, *et al.*[26], evaluated the fly ash and lime dosages selected for Lake Charles East by means of laboratory jar tests, *in situ* column studies, and algal assays.

The actual properties of a given fly ash depend upon its source, the type of coal burned, the types of combustion and ash collection equipment employed, and the method of storage or disposal. Before a fly ash is used in lake restoration its potential deleterious effects should be methodically considered and weighed against potential benefits of accelerated lake recovery.

SECTION V
SITE DESCRIPTION

GENERAL

The subject area is Lake Charles East, located approximately 8 km north of the town of Angola in Steuben County, Indiana. Figure 1 shows the lake, its drainage area, and associated relevant landmarks. Figure 2 is a bathymetric map of the eastern portion of the lake, which was treated with fly ash. Table 2 gives relevant data on the general features of the lake. Lake Charles is a shallow man-made lake which is fed by a small stream at its east end. There is also an outlet at the west end. Filling of the lake began in 1956 and was completed in mid-1958. In 1967 an interstate highway bridge (I-69) was constructed and put into service. This is shown in both Figures 1 and 2.

The land surrounding the lake has been subdivided, however, few houses have been built to date. The major influencing development has been a motel which occupies the eastern half of the lake. Beginning in August, 1968, the effluent from the motel's sewage treatment plant was allowed to run into the lake. Although the treatment plant brought about acceptable removals of suspended solids and dissolved organic carbon, phosphorus and nitrogen levels were consistently high and resulted in the deterioration of water quality in Lake Charles East. Abatement of this source of nutrients was achieved through the installation of a spray irrigation field south of the watershed (see Figure 1), in September, 1974.

The abatement of the major source of nutrients into Lake Charles East was

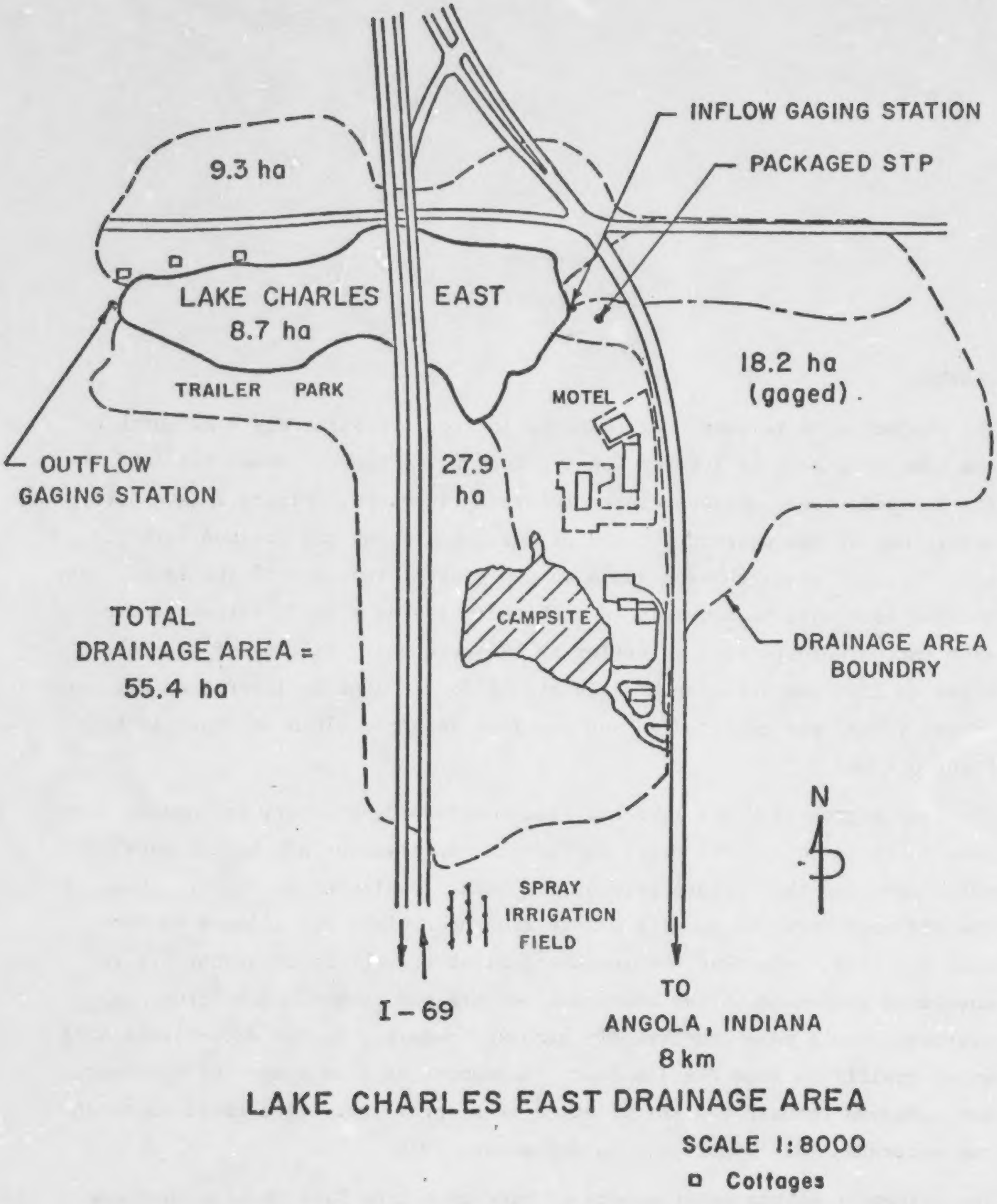
INFLOW GAGING STATION
PACKAGED STP
9.3 ha
LAKE CHARLES EAST
8.7 ha
TRAILER PARK
OUTFLOW GAGING STATION
MOTEL
18.2 ha (gaged)
27.9 ha
CAMPSITE
TOTAL DRAINAGE AREA = 55.4 ha
DRAINAGE AREA BOUNDRY
N
SPRAY IRRIGATION FIELD
I-69
TO ANGOLA, INDIANA 8 km
LAKE CHARLES EAST DRAINAGE AREA
SCALE 1:8000
Cottages

FIGURE 1: Lake Charles East Area with Important Landmarks

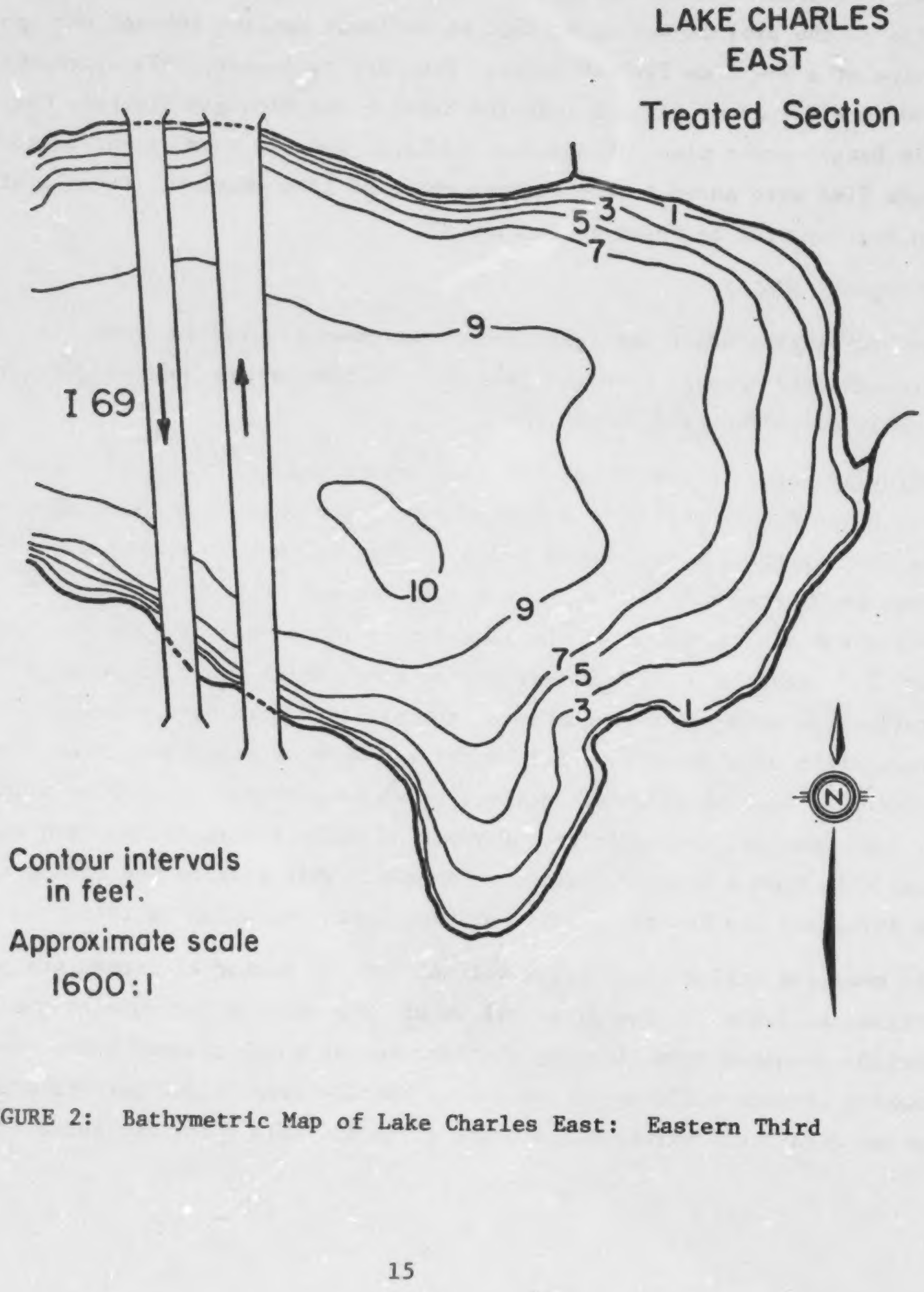
LAKE CHARLES
EAST
Treated Section
I 69
1
3
5
7
9
10
9
7
5
3
1
N
Contour intervals
in feet.
Approximate scale
1600:1

FIGURE 2: Bathymetric Map of Lake Charles East: Eastern Third

closely timed with the inception of this project. Intensive monitoring of physical, chemical, and biological parameters in Lake Charles began in mid-1974. It was felt at that time, and has subsequently been verified through measurements, that after abatement the sediments of Lake Charles represented a major source of nutrients at certain times during the year. The major focus of the project was an attempt at sediment sealing through the application of a 2 - 5 cm fly ash layer. From May to August, 1975 approximately 1400 metric tons of fly ash from the Indiana and Michigan Electric Company's Twin Branch power plant (Mishawaka, Indiana) and 270 tons of agricultural grade lime were added to the eastern third of Lake Charles. Details of the application will be found in Section VI.

HYDROLOGIC BUDGET

The hydrologic budget for Lake Charles was investigated in order to: (1) determine the seepage from the lake and (2) make an estimate of average hydraulic detention time in the lake.

Hydraulic Data. A portion of the lake inflow and the total lake outflow were gaged from June, 1975 to September, 1976 with some periods of no record. The instrumentation used consisted of triangular-notch weirs with continuous stage recorders installed on both the inflow and outflow streams. Precipitation was also measured at the lake during the summer of 1976 with a standard U.S. Weather Service non-recording gage, which was read weekly. Evaporation pan data and more complete precipitation data were necessary for computation of a detailed water budget and were obtained from a weather station at Prairie Heights, Indiana, which is approximately 20 km southwest of Lake Charles. In addition, 30 years of daily precipitation data were available from a weather station at Angola. This station was discontinued in 1974, and the Prairie Heights station began operating in 1973.

The measured inflow and outflow volumes for the period of record are summarized in Table 3. The large values of lake outflow for some of the gaging periods resulted from cleaning the lake outlet which clogged quite often, causing storage build-up in the lake. Monthly evaporation pan and precipitation data from Prairie Heights are given in Table 4 for the years 1974 to

1976. Normal monthly precipitation (30-yr. average) for Angola is also included in the table.

Average annual precipitation for the geographic location of Lake Charles is 88.9 cm (35.0 inches), and average annual lake evaporation is 80 cm (31.5 inches)[27]. Pan evaporation has an average annual value of 104.1 cm (41.0 inches), which gives an annual pan coefficient of 0.77. Lake Charles is located in the St. Joseph River Basin which has an average annual runoff of 24.1 cm (9.5 inches); however, approximately 80 percent of this runoff is due to groundwater seepage into the larger streams[28].

Lake and Watershed Characteristics. Watershed surface deposits are composed primarily of glacial till. The Soil Conservation Service hydrologic classification of the surficial soils is class B, which indicates moderate infiltration rates with moderately fine to moderately coarse soil texture. Total watershed relief is approximately 33.5 m (110 ft.) from elevation 311 m to 344 m (1020 ft. to 1130 ft.) above MSL. The watershed topography is characterized by moderately steep slopes with some depressions.

The watershed drainage areas for the lake, both gaged and ungaged, are shown in Figure 1 and summarized in Table 5. The percent of impervious area is also given in the table. Watershed cover on the pervious area consists of heavy grass and weeds with some forest. There is no agricultural cultivation on the watershed.

The Lake Charles watershed is underlain by 90 to 120 m (300 to 400 ft.) of glacial drift, which contains numerous sand and gravel deposits that are sources of groundwater. Bedrock is gray shale at elevation 213 m (700 ft.)[101]. The motel located adjacent to Lake Charles has two wells for which the driller's logs are given in Table 6. Approximate elevations of the wells from a USGS topographic map indicate that the top of the water-bearing gravel formation is located 6 to 15 m (20 to 50 ft.) below the elevation of the bottom of Lake Charles with static water levels very near the lake surface. It must be pointed out, however, that the gravel formation is a confined aquifer and thus does not seem to be feeding Lake Charles with inflow seepage. Water surface elevation for the lake is 310.9 m (1020 ft.) at which the water surface area is 8.7 ha (21.4 acres). Total lake volume

TABLE 2

General Features of Lake Charles East

Surface Area (ha)	8.6
Max. Depth (m)	3.2
Mean Depth (m)	2.0
Drainage Area (ha)	55.4
Volume (m^3)	195,000
Average Hydraulic Retention Time (years)	2.4
Alkalinity (mg/ℓ as $CaCO_3$)	60 - 235[+]
pH	7.0 - 10.0[+]

[+] High and Low Values Prior to Treatment is 195,000 m^3 (158 acre-feet).

Seepage Determination. The time periods from April 14 to May 13, 1976 and July 21 to August 18, 1976 were selected to obtain a detailed water budget for Lake Charles so that a seepage determination could be made. The general storage equation was utilized in the following form:

$$\Delta S = I_g + I_u + P - E - O - O_S \qquad \text{V-1}$$

in which ΔS is the change in storage volume for the time period (positive for increasing storage with time), I_g is gaged inflow volume, I_u is ungaged inflow volume, P is the precipitation volume falling on the lake surface, E is lake evaporation, O is lake outflow through the outlet structure, and O_s is seepage (positive out of the lake). All quantities in equation 1 were either measured or calculated except for O_s, which was then solved for to obtain a seepage estimate.

The quantities ΔS, I_g, and O were measured at the lake site for the two time periods specified above. Daily precipitation values from Prairie Heights were obtained for the first time period in April and May, while precipitation data at the lake site was available for the second time period.

Ungaged inflow to the lake was calculated from the SCS equation:

$$Q = \frac{\left(P - \frac{200}{CN} + 2\right)^2}{P + \frac{800}{CN} - 8} \qquad \text{V-2}$$

in which P is storm precipitation, CN is the runoff curve number, and Q is the volume of storm runoff. Runoff curve numbers were selected based on the watershed cover and antecedent moisture conditions. Volumes of runoff were then accumulated for the time periods under consideration for each storm occurrence. Equation 2 has been used extensively by the U.S. Soil Conservation Service for ungaged drainage areas. It is based on the assumption that the ratio of actual infiltration to potential infiltration is

TABLE 3

Measured Lake Inflow and Outflow

	Inflow		Outflow	
Dates	m^3	cm*	m^3	cm*
6/4-6/12, 1975	1628	1.88	-	-
6/12-6/20	2245	2.59	-	-
6/20-6/27	370	0.43	-	-
6/27-7/3	86	0.10	86	0.10
7/3-7/10	111	0.13	765	0.89
7/10-7/18	111	0.13	136	0.15
7/18-8/1	0	0.00	62	0.08
8/1-8/12	7981	9.22	86	0.10
8/12-8/20	-	-	62	0.08
8/20-8/28	839	0.97	-	-
11/12-11/20	49	0.05	222	0.25
11/20-11/28	259	0.30	1160	1.35
11/28-12/5	1468	1.70	1628	1.88
12/5-12/13	703	0.81	1382	1.60
3/9-4/14, 1976	3675	4.24	54447	62.87
4/14-5/13	5415	6.25	1850	2.13
5/13-6/8	1567	1.80	-	-
6/8-7/8	1258	1.45	-	-
7/12-8/18	530	0.61	10694	12.34

*cm of water on the lake surface with area 8.7 ha (21.4-ac)
-missing records

TABLE 4

Precipitation Records

	Prairie Hts. 1974		Prairie Hts. 1975		Prairie Hts. 1976		Angola Normal*
	Precipitation cm	Pan Evaporation cm	Precipitation cm	Pan Evaporation cm	Precipitation cm	Pan Evaporation cm	Precipitation cm
Jan.	6.91	-	4.95	-	2.01	-	5.13
Feb.	3.91	-	5.26	-	5.66	-	4.52
March	7.04	-	2.87	-	7.32	-	7.04
April	4.42	5.31	11.58	0.84	8.10	9.02	9.45
May	13.66	11.07	12.14	17.07	5.99	14.15	9.25
June	8.66	13.89	19.10	14.65	14.48	19.00	10.08
July	1.73	23.19	3.94	18.29	16.76	19.30	8.10
Aug.	5.30	16.81	27.43	11.73			8.15
Sept.	9.96	11.46	2.92	8.74			8.30
Oct.	2.72	7.90	2.97	9.53			7.06
Nov.	5.97	0.58	11.43	3.07			6.81
Dec.	4.52	-	4.47	-			5.08
TOTAL	74.80	90.22	109.07	83.92			88.98

*30-year average

TABLE 5

Lake Charles Watershed Drainage Area

	Drainage Area, acres	Impervious Area, %
Gaged Area	50 (20.3 ha)	10
Ungaged Area	87 (35.3 ha)	15
Total	137 (55.6 ha)	13

TABLE 6

Motel Well Logs

Well No. 1* - Static Water Level 77 ft. (23.5 m)

Formation	Depth, ft. (m)
Yellow clay	0-15 (0-4.6)
Gray clay	15-85 (4.6-25.9)
Brown gravel	85-90 (25.9-27.4)
Gray Clay	90-110 (27.4-33.5)
Water gravel	110-137 (33.5-41.8)

Well No. 2** - Static Water Level 110 ft. (33.5m)

Formation	Depth, ft. (m)
Surface	0.5 (.15)
Gravely clay	5-90 (1.5-27.5)
Gray clay	90-150 (27.5-45.7)
Clean gravel	170-190 (51.8-57.9)

*Approximate surface elevation 1100 ft. (335.3m)
**Approximate surface elevation 1130 ft. (344.4m)

equal to the ratio of direct runoff to potential runoff (storm rainfall minus initial abstraction).

Lake evaporation was computed from daily pan evaporation data collected at Prairie Heights. The daily pan coefficient was computed from the daily wind movement, air temperature, and pan water temperature according to a method developed by Kohler for the U.S. Weather Bureau[29].

The results of the water budget analyses are presented in Table 7. The seepage estimates are reported as depth of water on the lake surface with an area of 8.7 ha (21.4 acres). The values for seepage are 0.76 cm (0.3 in.) for the first time period of 29 days and 0.25 cm (0.1 in.) for the second period of 28 days. Based on these results and taking into account the possible errors involved in obtaining them, it is concluded that seepage from the lake is less than 1.3 cm (0.5 in.)/month and can be neglected. In detention time on an average annual basis, a seepage from the lake of 1.3 cm/month produces a maximum possible relative error in the detention time of 20 percent. This is considered acceptable in view of the gross nature of a detention time which is based on the assumption of a constant and continuous lake outflow.

Detention Time. An average annual hydraulic detention time in the lake was computed from the total storage of 195,000 m^3 (158 ac-ft.) and an average annual outflow determined from:

$$O = I + P - E \qquad \text{V-3}$$

where I is the annual surface inflow to the lake, P is annual precipitation, and E is annual lake evaporation. The annual change in storage has not been included in equation 3 since it is very small compared to the average annual outflow. The seepage has also been left out of equation 3 for reasons above. Average annual values for P and E, which were given earlier in the report, were used in equation 3. Surface inflow I was estimated by applying a runoff coefficient to the annual precipitation. Although this is a crude method of estimating runoff that should never be applied on a storm by storm basis, it is acceptable for determining seasonal or annual runoff since the effects of antecedent moisture conditions and other storm factors are damped out over longer time periods.

Estimation of the runoff coefficient was guided by the available inflow and precipitation data for the gaged portion of the Lake Charles watershed. A straight line was fitted to this data by the method of least squares to obtain an average monthly runoff coefficient of 0.18 for the months of April through November. Calculating a weighted runoff coefficient from published values gave a slightly different result. A coefficient of 0.05 was selected for the pervious portion of the watershed based on annual water yield data for similar experimental watersheds in Coshocton, Ohio[30]. Choosing a coefficient of 0.85 for the impervious areas and weighting the coefficients according to the presence of pervious and impervious drainage areas found in Table 5 resulted in a value of 0.15 for the annual runoff coefficient. This value is approximately 15 per cent less than the growing season value of 0.18. Inasmuch as the coefficient of 0.15 more closely represents annual runoff rather than seasonal runoff, it was used in the calculation of detention times.

The result for the average annual lake detention time was 2.4 years. Detention times were also determined for the calendar years of 1974 and 1975 to be 2.9 years and 1.5 years, respectively. Finally, a frequency analysis was performed on the lake detention times calculated from annual precipitation and evaporation data. The results are shown in Table 8, in which the detention times given can be expected to be equalled or exceeded on the average of once every N years, where N is the recurrence interval. A normal distribution was fitted to the data to obtain the recurrence intervals in the table.

WATER QUALITY DATA

Lake Charles East is a hard water lake characterized by high alkalinities and alkaline pH values. The water quality parameter of greatest concern is phosphorus. Table 9 gives the external phosphorus budget as determined by calculation, measurement, and in some cases estimates for the period 1959

TABLE 7

Seepage Estimates from Water Budget+

	Time Periods	
	4/14 - 5/13, 1976	7/21 - 8/10, 1976
Change in Storage, ΔS	1.0*	-5.0*
Gaged Inflow, I_g	2.5	0.4
Ungaged Inflow, I_u	0.8	0.2
Precipitation, P	4.0	4.4
Evaporation, E	5.2	5.0
Outflow, O	0.8	4.9
Seepage, O_s	0.3	0.1

*All values reported as in. of water on lake surface with area of 21.4 ac.

+Multiply inches by 2.54 to obtain centimeters.

TABLE 8

Detention Time Recurrence Intervals

Recurrence Interval, yrs.	Detention Time, yrs.
1.25	1.6
2.	2.2
5.	2.8
10.	3.1
25.	3.5

TABLE 9

Yearly Total Phosphorus Inputs for Lake Charles East: 1959-1978 (gm P/year)

1	2	3	4	5	6	7	8	9
	Runoff							
Year	Pervious	Impervious	Precipitation & Dustfall	Wastewater Input	Forest Litter	Total P Input	Δ P Storage	Total P Outflow
1959 *	1110	810	950	0	1000	3870	0	2300
1960 *	890	640	800	0	1000	3330	0	1570
1961 *	1030	750	860	0	1000	3640	0	2170
1962 *	745	540	750	0	1000	3035	0	818
1963 *	705	520	750	0	1000	2975	0	541
1964 *	810	590	780	0	1000	3180	0	1110
1965 *	1230	890	1000	0	1000	4120	0	2490
1966 *	1060	1400	850	0	1000	4310	0	2380
1967 *	1110	1480	900	0	1000	4490	0	2720
1968 *	1220	1670	950	124000	1000	128840	+62700	38200
1969 *	1020	1670	820	323000	1000	327510	+127000	98200
1970 *	1020	1670	820	323000	1000	327510	+72500	130000
1971 *	765	1250	750	323000	1000	333000	+39200	91200
1972 *	1120	1840	920	194000	1000	198880	-21600	164000
1973	1070	1750	860	204020	1000	208700	-92120	134600
1974	900	1470	780	139550	1000	143700	-60760	59900
1975	1320	2140	1000	2240	1000	7700	-39200	66000
1976	1020	1670	880	7530	1000	12110	-25500	26400
1977	1070	1750	880	0	1000	4700	-17640	19200
1978*	1070	1750	880	0	1000	4700	- 3920	19000

*Calculated

to 1978. It should be noted that these figures do not include any contribution from the sediments. This will be treated in Sections VII and VIII. The average areal phosphorus loading for the year preceding abatement was 2.43 g-P/m^2/yr. After abatement (1975-1978) the phosphorus loading was 0.25 g-P/m^2/yr., a reduction of 90%. Various yearly loadings for Lake Charles East are located on Figure 3, which is a diagram of Vollenweiders phosphorus loading curves[31]. The peak total phosphorus and soluble ortho phosphate levels measured during 1974 were 970 μg-P/ℓ and 714 μg-P/ℓ, respectively, although it appears that phosphorus concentrations were even higher prior to the Indiana phosphate ban on detergents enacted in 1972. This is reflected in the data of Table 9.

Available data on phosphorus concentration prior to the initiation of this study are given in Table 10.

Nitrogen

Ammonia levels in Lake Charles East are generally high. Most values are consistently in the range of 0.2 to 0.5 mg/ℓ as N. It is produced primarily through the deamination of organic nitrogen. These high values probably play a role in the incidence of fish kills which have been noted previously in the lake. Oxidized forms of nitrogen (nitrate + nitrite) display very low levels, especially during the summer season.

Suspended Solids

If a suspended solids range of 5-20 mg/ℓ is representative of bloom conditions, as suggested by some[34], then it is clear that Lake Charles East was in a continued state of bloom prior to abatement and treatment. Secchi disc transparencies reflect this also and in fact, a general relationship between secchi disc and suspended solids can be seen in Figure 4. The data on this log-log plot tend to agree with the exponential relationship for absorption of light by water,

$$I_z = I_o e^{-\eta z} \qquad \text{V-4}$$

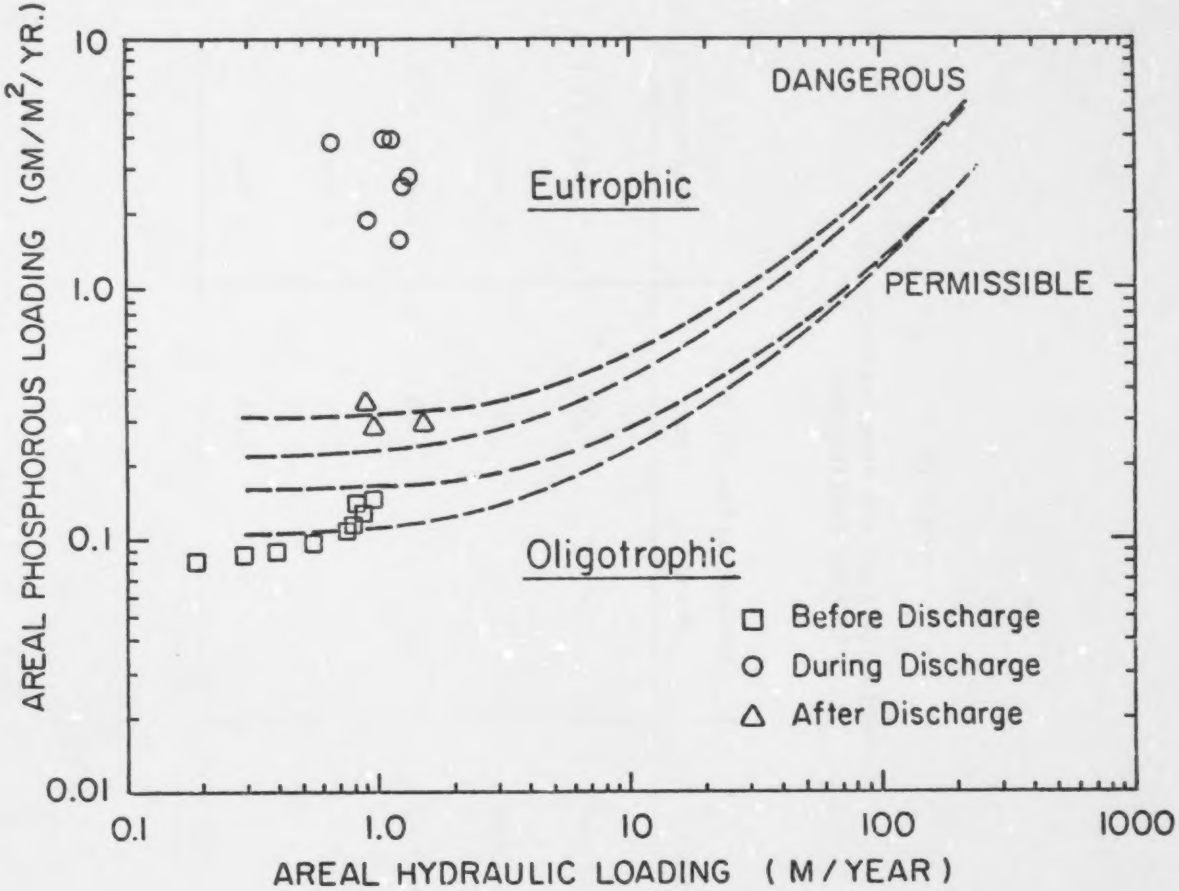

FIGURE 3: Depiction of Phosphorus Loading for Lake Charles East During Various Stages as Compared with Vollenweiders Loading Curves.

TABLE 10

Previously Reported Phosphorus Levels in Lake Charles East

Date	Concentration (mg-P/ℓ)		Reference
	Ortho	Total	
11-18-67	-	0.03	surface (32)
7-16-69	-	0.7	8 feet (33)
6-20-72	-	1.99	surface (32)
		1.34	5 feet
		1.24	8 feet

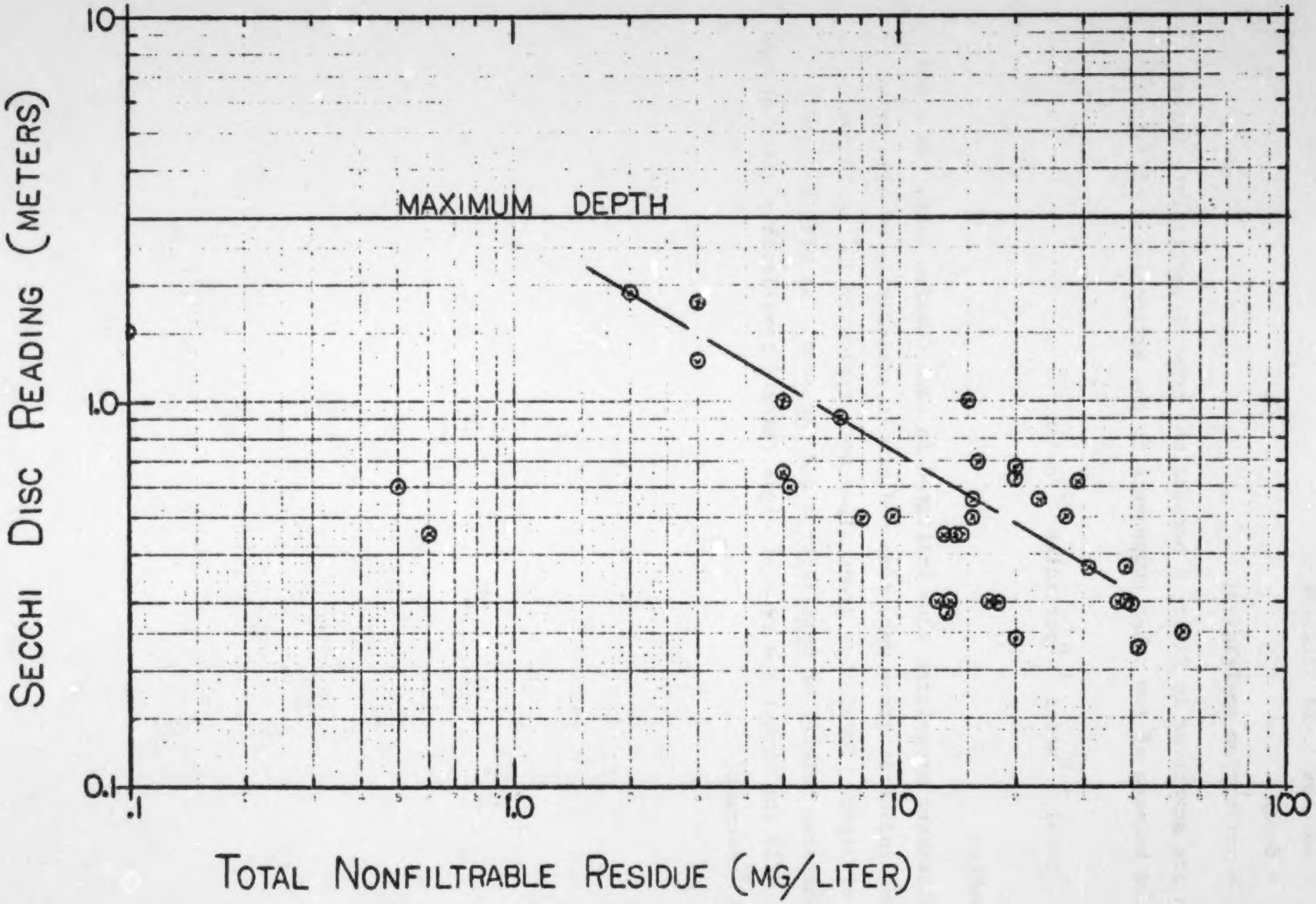

Figure 4: Secchi Disc Reading Vs. Nonfiltrable Residue.

where I_z = light intensity at depth z,

I_o = surface light intensity,

z = depth

and η = extinction coefficient.

The data are scattered in Figure 4 because the type of particulate matter varied and because of two other components in the extinction coefficient,

$$\eta_{total} = \eta_{water} + \eta_{particles} + \eta_{color}.$$

Phytoplankton

Before abatement procedures were initiated in Lake Charles East, the bloom conditions indicated above were composed almost exclusively of blue green species of algae. Total blue green algae were typically 10^4 to 10^5/mℓ. These consisted chiefly of Oscillatoria and Anabaena. As will be seen (Section VII) the number and type of algae changed considerably after abatement and treatment.

SECTION VI

METHODOLOGY OF APPLICATION OF FLY ASH TO LAKE CHARLES EAST

GENERAL

Experience with fly ash at Lake Charles East points out the importance of material specifications, accessibility to the site, weather conditions, and bulk material handling methods for projects of this type.

The fly ash was obtained from the Indiana and Michigan Electric Company's Twin Branch Generating Station in Mishawaka, Indiana.

An analysis of the fly ash is given in Table 11. Because of the low calcium oxide content, a supplemental quantity of lime (270 tons) was added to the lake along with the fly ash. Previous studies[25] had indicated the importance of the lime content and at the initiation of the study it was believed to be an essential component.

DELIVERY

The fly ash pond at Mishawaka had dried sufficiently since I&M had switched from coal to oil in 1973. Heavy machinery and trucks were able to drive on the fly ash. A thin layer of bottom ash, placed to hold down dust, was removed from the surface of the fly ash, and a drag line was used to excavate and pile up approximately 2000 tons of the fly ash. An articulated front end loader was used to load the fly ash into a succession of 30 cubic yard tandem (dual axle) trailer dump trucks. The loads were weighed and trucked approximately 65 miles to the lake site.

In order to achieve a uniform 4-5 centimeter layer of ash on the lake sediments (previous studies by Yaksich[25] had indicated a 2 cm layer was needed at

TABLE 11

Chemical Composition of Fly Ash Added to Sediments of Lake Charles East (composite sample)

Component	Concentration
CaO	4.2 %
As_2O_3	42.2 %
Fe_2O_3	32.9 %
$Si\ O_2$	20.6 %
As	80 μg/g
Cd	8 μg/g
Cr	90 μg/g
Cu	48 μg/g
Hg	20 μg/g
Mn	225 μg/g
Ni	220 μg/g
Pb	85 μg/g
Zn	330 μg/g

minimum) it was necessary to load fly ash to the treated section of the lake at the rate of approximately 500 tons/ha (200 tons/acre). A total of 1973 tons of fly ash were excavated and delivered to the lake. The average moisture content was 22.8% leaving 1523 tons of dry ash for application. The first series of trucks used tarps to prevent fly ash from blowing on the highway, however, it was soon discovered that the tarps were unnecessary. Apparently, the high moisture content of fly ash freshly removed from the pond helped prevent a dust problem.

The open dump trucks were originally chosen over bulk material, cement-type trucks because they were cheaper, in greater supply, and more suitable for ponded ash. Completely enclosed trailers and bulk storage hoppers with dust control devices would probably be required for dry fly ash delivery and storage.

Site conditions, such as accessibility by road, soil moisture content, and levelness of the ground, were extremely important in unloading the fly ash. Many of the dump trucks also became stuck while backing into the fly ash stockpile area and they had to be emptied in place. A small bulldozer was used in a dual role: to pull out the trucks and to pile up the fly ash so that more trucks could dump. The trucks also had to be emptied on a relatively level area. Otherwise, they could have easily tipped over when the loads were raised to dump.

FLY ASH AND LIME APPLICATION METHODS AND EQUIPMENT

The fly ash and lime were applied to the treated side of Lake Charles East by means of a piped water slurry system. The work site and the fly ash and lime stockpiles were on a level area about 8 m (26 ft.) above the lake surface elevation at the northeast corner of the lake (Figure 1).

A 10.2 cm (4 in) variable-speed, 2.26 m^3/min. (600 gpm) capacity trash pump was used to pump screened lake water out of the lake, through a 10 cm diameter canvas discharge hose, and into a 10 cm diameter black steel spray bar 6.3 m (20.6 ft) long. The spray bar contained 80-0.80 cm diameter spray holes at 1.2 cm (3 in) on centers in two rows 3.9 m (12.8 ft) long. At full pump capacity up to 28.4 ℓ/min. (7.5 gpm) would have come out of each spray bar hole at about 9.1 m/sec (30 fps) depending on the amount of debris clogging some of the holes. Actual pump discharge was 1.7 - 2.1 m^3/min. (400-500 gpm) due to head and friction losses and pump control to match the capacity of the slurry discharge line. The scouring action of the water as it left the spray bar was sufficient to suspend the fly ash particles, forming a concentrated aqueous slurry of about 15% by volume. This slurry was collected via a system of ditches and channeled into a 20.3 cm (8 inch) corrugated plastic irrigation pipe which carried the slurry to the lake by gravity flow. This pipe was chosen because it is flexible, lightweight, easy to connect and disconnect in the field, slightly buoyant and relatively inexpensive.

During operation actual flow velocities in the slurry discharge line varied from about 0.3 to 0.6 m/sec (1 to 2 fps) depending on pump discharge rate, cross-sectional area of flows, and wall effects of corrugations. A velocity of 0.3 m/sec was more than sufficient to keep individual fly ash particles suspended, but the fly ash was contaminated with numerous larger and heavier particles which tended to settle out in the pipe and cause sinking and blockages.

There were numerous types of contamination in the fly ash from the following sources:

1. Sand from the dike around the I&M fly ash pond.
2. Bottom ash from the dust control layer on the fly ash pond.
3. Large chunks of very hard ash material, some measuring 20 to 30 cm (8-12 in) across, caused either by high concentrations of calcium, iron, or other cementitious material or high pressure in certain layers of fly ash in the pond.
4. Organic matter, such as sticks and leaves from the unimproved bottom of the fly ash pond, weeds from the stockpile site, and clayey sod turned up by the bulldozer.
5. Sand and stones from the gravel which was spread at the stockpile site to improve bearing capacity and traction for trucks and the front end loader.

The contaminants, especially the large chunks of slag and heavy gravel particles, were a major source of difficulty throughout the summer. Without them, the slurry discharge system would have worked with far less irritation, the pipe probably would not have plugged and broken so often, and much more fly ash would have been available to treat the lake. The contaminants appeared to compose about 5 to 10% of the fly ash stockpile. About 200 tons of heavy contaminated material was discarded at the end of the project.

Although fly ash contaminants may not be intrinsically harmful in lake treatment, they do reduce the amount of effective fly ash, and organic material may contribute to the amount of floating debris and sources of nutrients. In retrospect, this particular application method might have worked better if the slurry had been directed into a sump and pumped out into the lake by another pump and hose. Another possible application technique on larger projects would be to push the fly ash into a large stockpile at the edge of a lake where a small dredge could scoop it and pump it out for distribution. Still another approach would be to obtain cleaner fly ash by using dry fly ash, improved ponding areas, or paved stockpile areas.

The corrugated plastic pipe used to carry the slurry to the lake required additional floatation in the form of air-filled, 12.7 cm (5 in) corrugated plastic pipe with ends capped. This was tied to the slurry discharge line to help prevent it from sinking from the weight of fly ash in the slurry and possible sediment build-up in the pipe. Air-filled pipe was preferred over other common floatation devices, such as metal drums, inner tubes, and styrofoam, because it was less expensive and it allowed continuous floatation along the length of the slurry discharge line.

The slurry discharge line was required to float because it was swept back and forth across the water to distribute fly ash and lime over the treated portion of the lake. It was impossible to move the pipe if a section sank to the bottom because the low point rapidly filled up with heavier particles from the fly ash slurry, especially when the flow rate and scour velocity declined or stopped.

The fly ash application method was similar to a dredging operation in reverse. The source of slurry material was at a fixed point, the fly ash pile, while the discharge end moved across the lake distributing fly ash which sank to the bottom. A dredge would normally move across the lake, taking sedimented material from the bottom and pumping it in slurry form to a fixed spoil location, either on land or in deeper water. One advantage of dredging is that either a boat or the dredge itself provides propulsion to pull the pipeline across the water and suction tends to position it.

In this project, however, the available pontoon boat with a 10 horsepower outboard motor was awkward and imprecise in positioning the end of the slurry discharge line. A flow of 2.12 m^3/min (500 gpm) exiting at an average velocity of .46 m/sec (1.5 fps) causes only 12.3 kg of thrust, but much higher drag forces were encountered in pulling the pipe back and forth across the water. Better positioning was obtained by stretching a 1.3 cm (0.5 in) Manila rope across the lake at 3.9 m (10 ft) intervals, attaching the end of the slurry discharge line to a small raft, and manually pulling the raft across the lake with a rope. This required frequent work stoppages to change the rope, but it allowed much more precise control of fly ash application because the same areas were not treated twice, and the depth of the fly ash layer could be controlled by adjusting the rate that the raft moved across the lake. The depth of the fly ash layer in place was measured by core samples.

RESULTS OF APPLICATION

Fly ash and lime were applied to the eastern third of Lake Charles East during the period May 10 to August 5, 1975. The total area covered was approximately 3 hectares (7.4 acres). The approximate area covered during the application period are shown in Figure 5. Figure 6 shows isopleths of fly ash depth above the sediments (in centimeters) as determined by a coring survey made in the spring of 1976. One difficulty which became apparent as a result of the survey was the continued deposition of organic detritus in the lake resulting in the establishment of a small but very noticeable (about 1 cm) layer above the fly ash. The effects of this will be discussed in Sections VII and VIII.

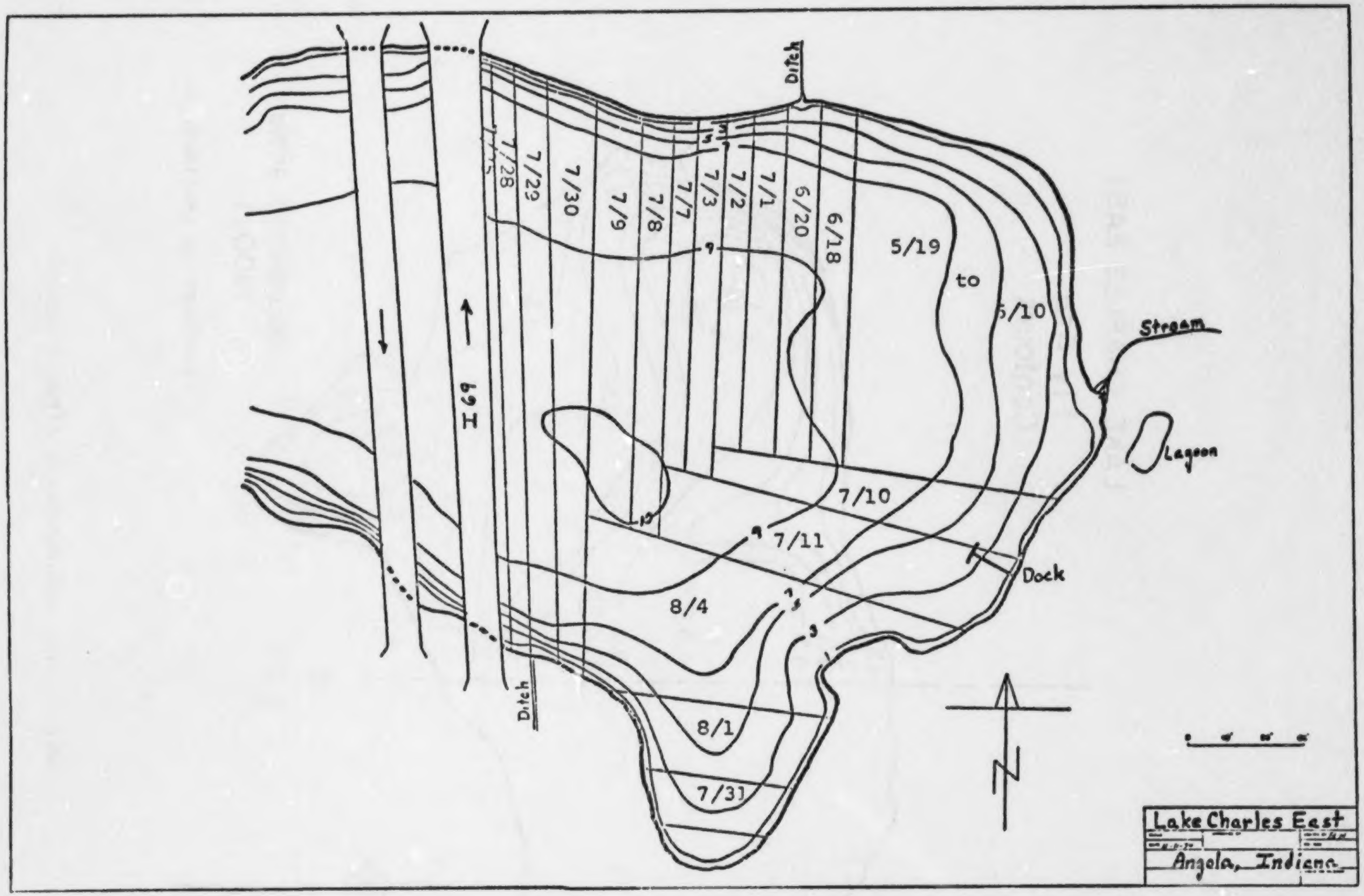
Ditch
7/28
7/29
7/30
7/9
7/8
7/7
7/3
7/2
7/1
6/20
6/18
5/19
to
6/10
Stream
Lagoon
7/10
7/11
Dock
8/4
8/1
7/31
Ditch
I 69
Lake Charles East
Angola, Indiana

FIGURE 5: Treatment of Lake Charles East - Summer 1975
(numbers represent dates).

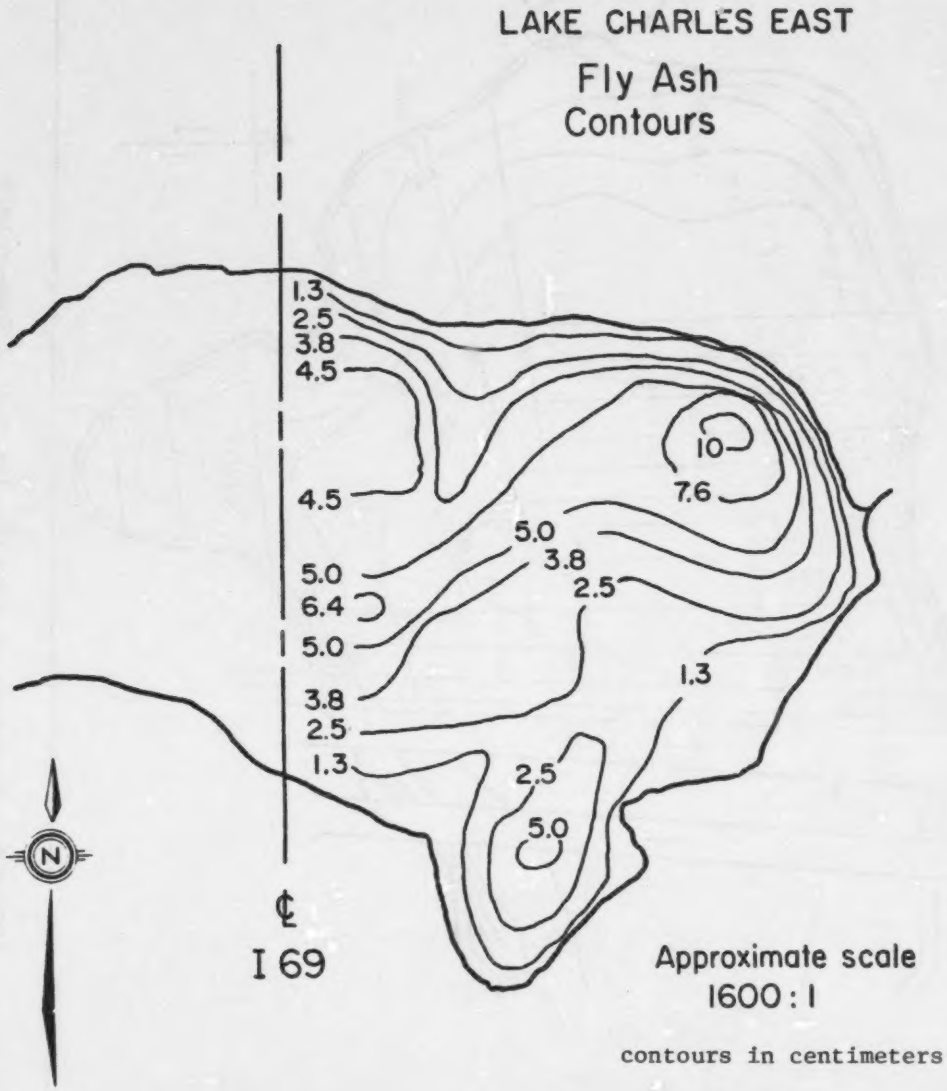
LAKE CHARLES EAST
Fly Ash
Contours
1.3
2.5
3.8
4.5
4.5
10
7.6
5.0
3.8
2.5
5.0
6.4
5.0
3.8
2.5
1.3
1.3
2.5
5.0
N
℄
I 69
Approximate scale
1600 : 1
contours in centimeters

FIGURE 6: Fly Ash Thickness After Treatment.

SECTION VII

RESULTS

WATER CHEMISTRY EFFECTS

The changes in water chemistry in Lake Charles East of greatest interest are for phosphorus and nitrogen. Both total and soluble reactive (filterable) phosphorus levels are shown in Figure 7 for the period of this study. Features of the data which are notable are the rapid decline in phosphorus after diversion of the wastewater, the low levels of soluble phosphorus during the summers during and after treatment, and the rise in phosphorus levels in the late summer and early autumn of each year after treatment. These will be discussed further in Section VIII.

The nitrogen levels in Lake Charles East are given in Figures 8 and 9 for ammonia and nitrite plus nitrate, respectively. Although the dynamics of the nitrogen cycle were not investigated specifically, a few observations can be made from the data. There is general agreement that most algae can utilize either ammonium or nitrate-nitrite as their source of inorganic nitrogen. Free ammonia, NH_3, can have a toxic effect, however. This fraction of the total increases with temperature and pH.

Figure 8 shows that average total ammonia concentrations in Lake Charles East are high. Ammonia is produced by the de-amination of organic nitrogen containing compounds. Free ammonia, NH_3, may have been a factor in fish kills which were observed on September 6, 1974 and August 8, 1975 (during lime addition) because both temperature and pH were high.

In Wisconsin lakes Sawyer found that nuisance algal blooms could be expected

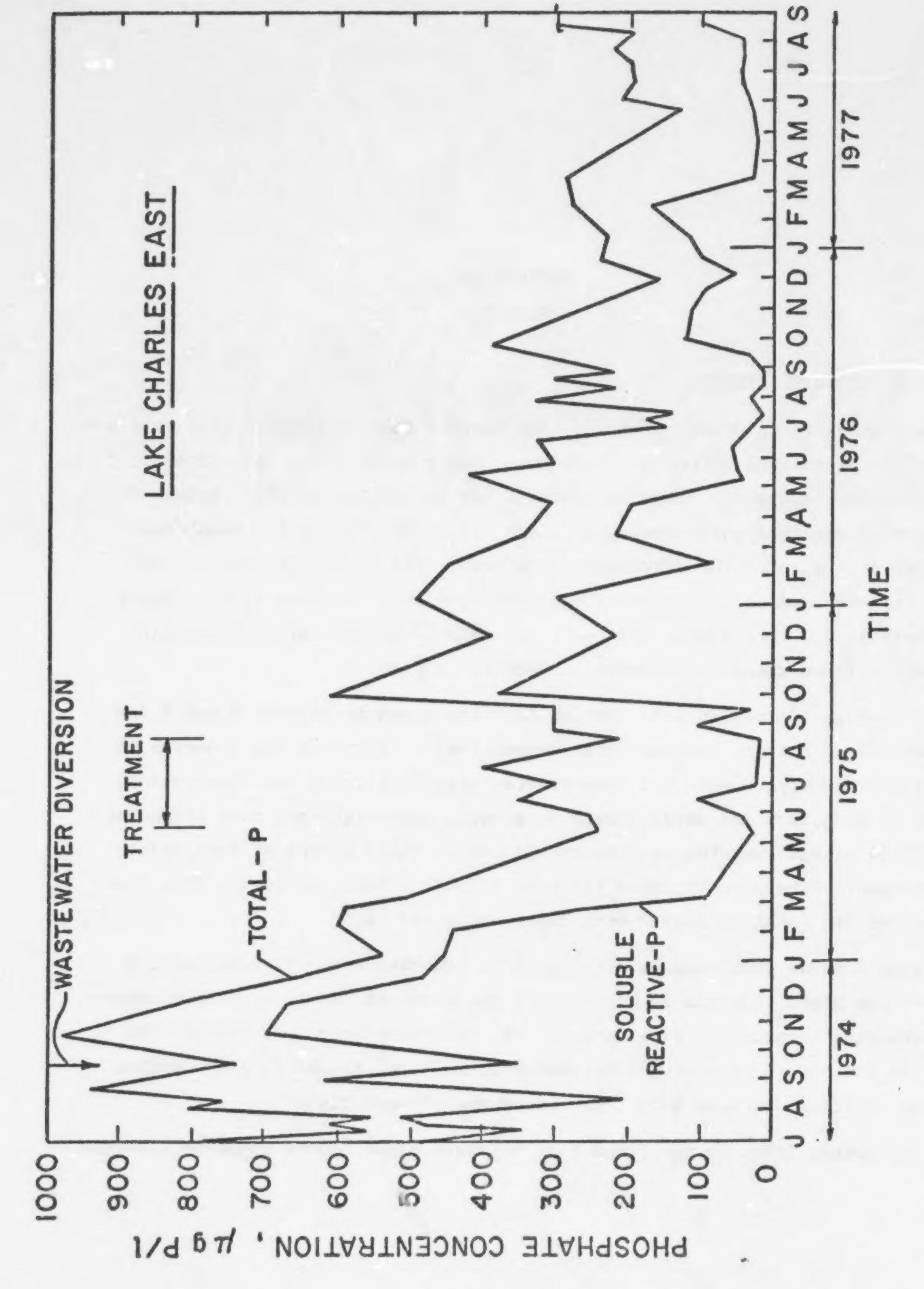

FIGURE 7: Total and Soluble Reactive Phosphorus in Lake Charles East.

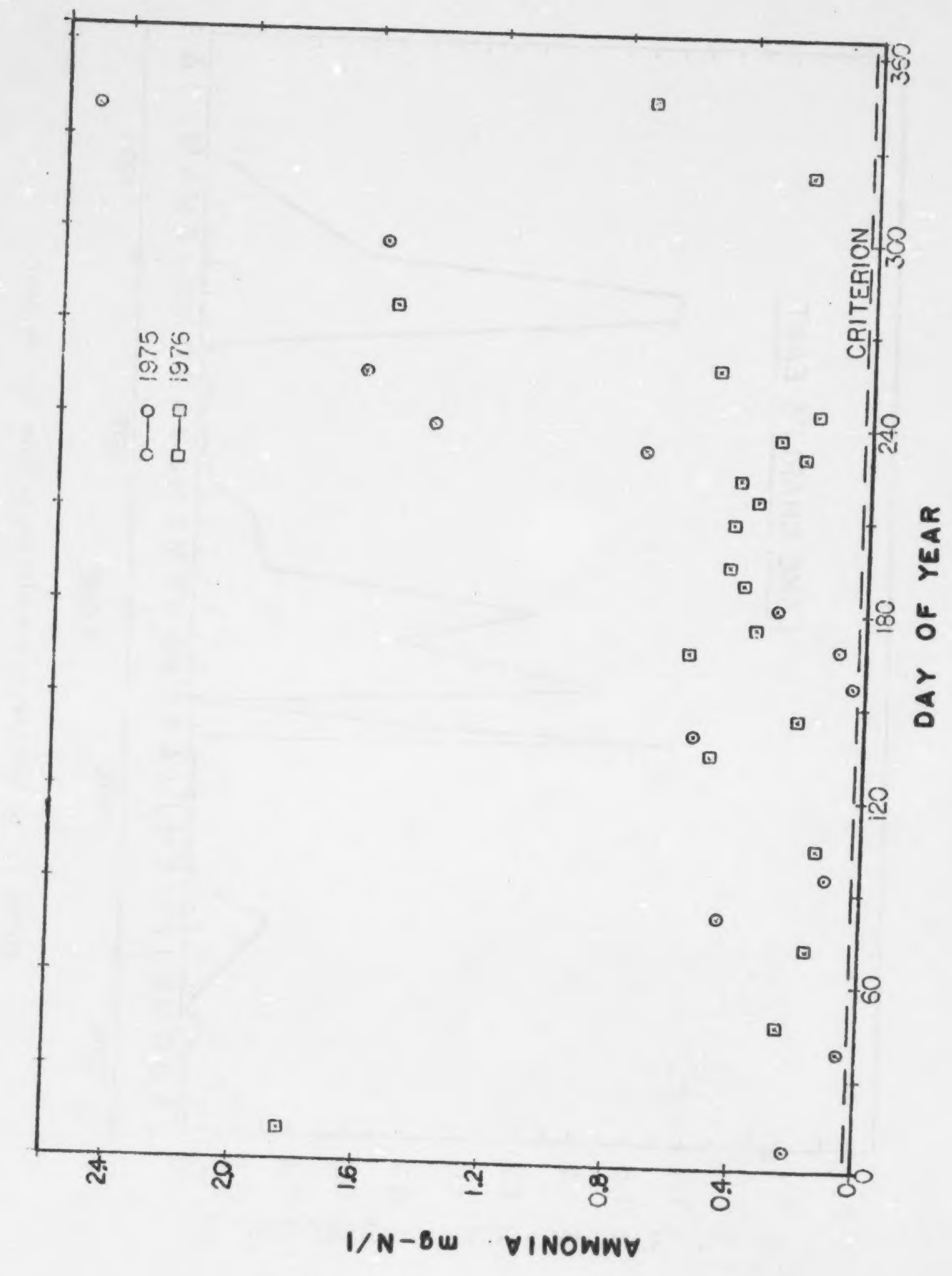

Figure 8: Average Total Ammonia Concentrations: 1975 - 1976.

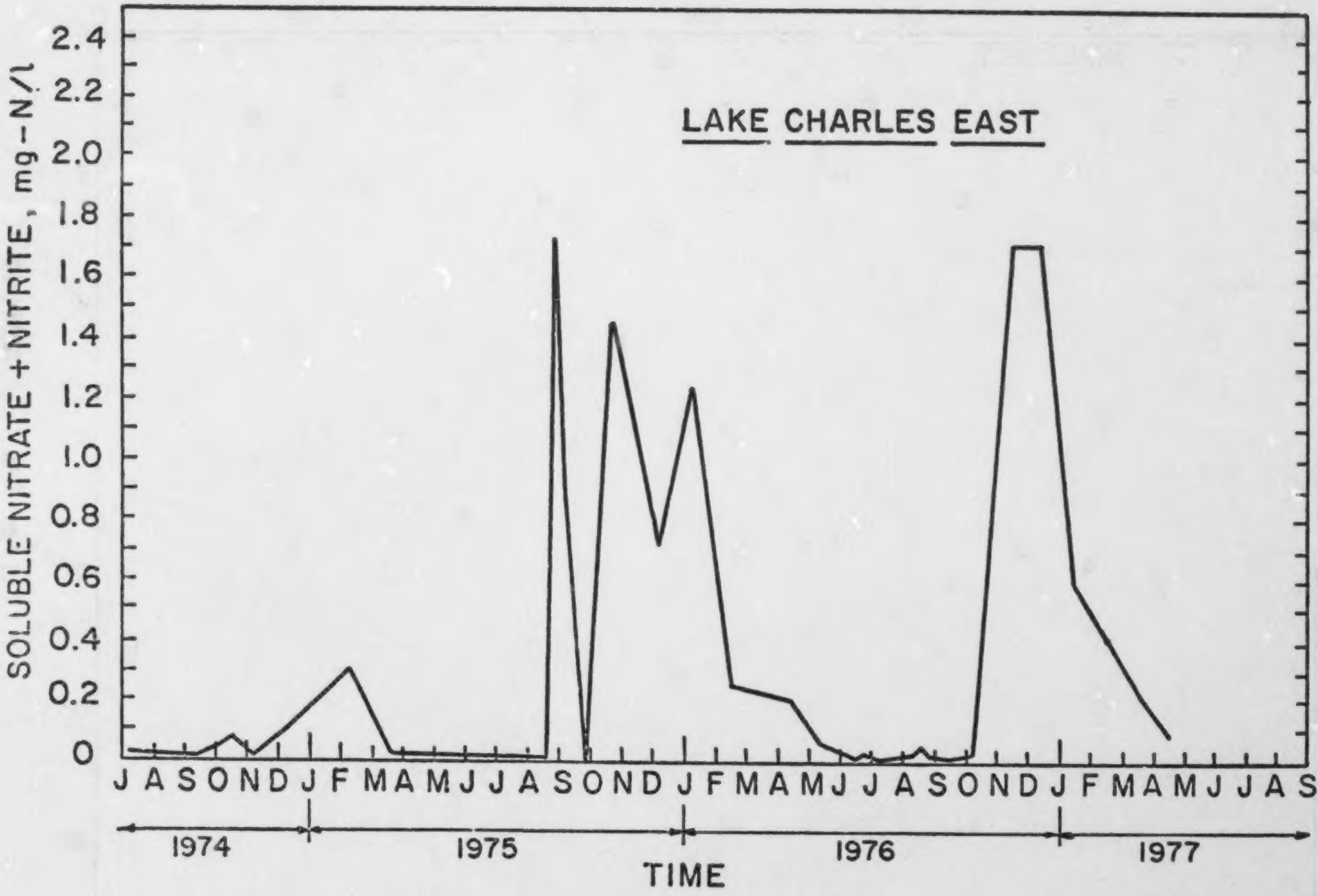

FIGURE 9: Soluble Nitrate + Nitrite in Lake Charles East.

when inorganic forms of nitrogen exceeded a critical level of 0.30 mg-N/ℓ in the spring. In Lake Charles East, the oxidized forms of nitrogen were often at growth limiting levels, especially in the summer months (Figure 9), but total inorganic nitrogen (i.e., including ammonia) often exceeded the critical level.

The filterable inorganic nitrogen:phosphorus ratios are plotted in Figure 10 as a means of estimating relative growth limitations. Various authors have indicated that a N:P ratio greater than 10-15 to 1 indicates a phosphorus limitation (or in some cases some other nutrient), while a lower ratio indicates nitrogen limitation. Lake Charles East can be considered as predominantly phosphorus limited during the summers of 1975 and 1976 as suggested by the phosphorus data of Figure 7. The dominance of nitrogen-fixing blue-green algal species in mid to late summer after diversion and treatment may be partially explained by noting a trend toward nitrogen limitations at those times of the year.

PHYTOPLANKTON EFFECTS

Relative phytoplankton composition is given graphically in Figure 11 for the period of this study. For convenience, the phytoplankton are divided into four functional groupings: blue-green, flagellates, green and diatoms. Biomass determinations as computed from phytoplankton counts in the lake are given in Figure 12[57]. The resultant water clarity, as determined by Secchi disk measurements, are given in Figure 13.

When biological sampling began on 26 June 1974, Lake Charles East was dominated by blue-green algae, primarily *Oscillatoria agardhii*. Algae present in the lake before treatment with fly ash are listed in Table 12.

In the first half of 1975 (that is, still prior to treatment) *Trachelomonas* became more abundant, reaching somewhat over 30% of total phytoplankton biomass. At about the end of January 1975, diatoms began increasing in importance as the *Trachelomonas* population ebbed. By April, both the greens and diatoms decreased in numbers and the blue-greens once again composed some 99% of the phytoplankton biomass. This condition existed up to, and during the actual fly ash treatment. Biomass fluctuated between values of 2-12

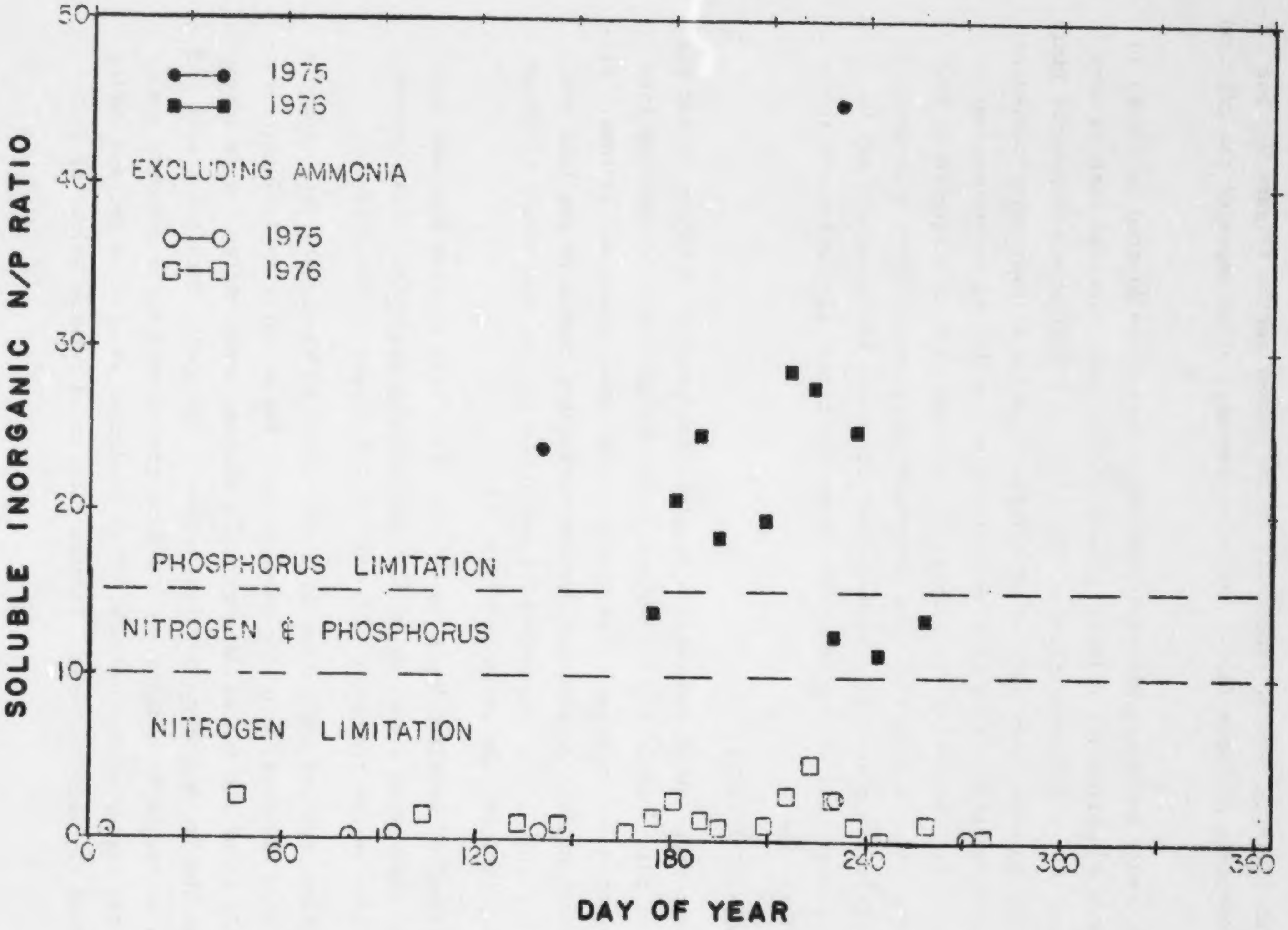

Figure 10: Filterable Inorganic Nitrogen: Phosphorus Ratios: 1975 - 1976.

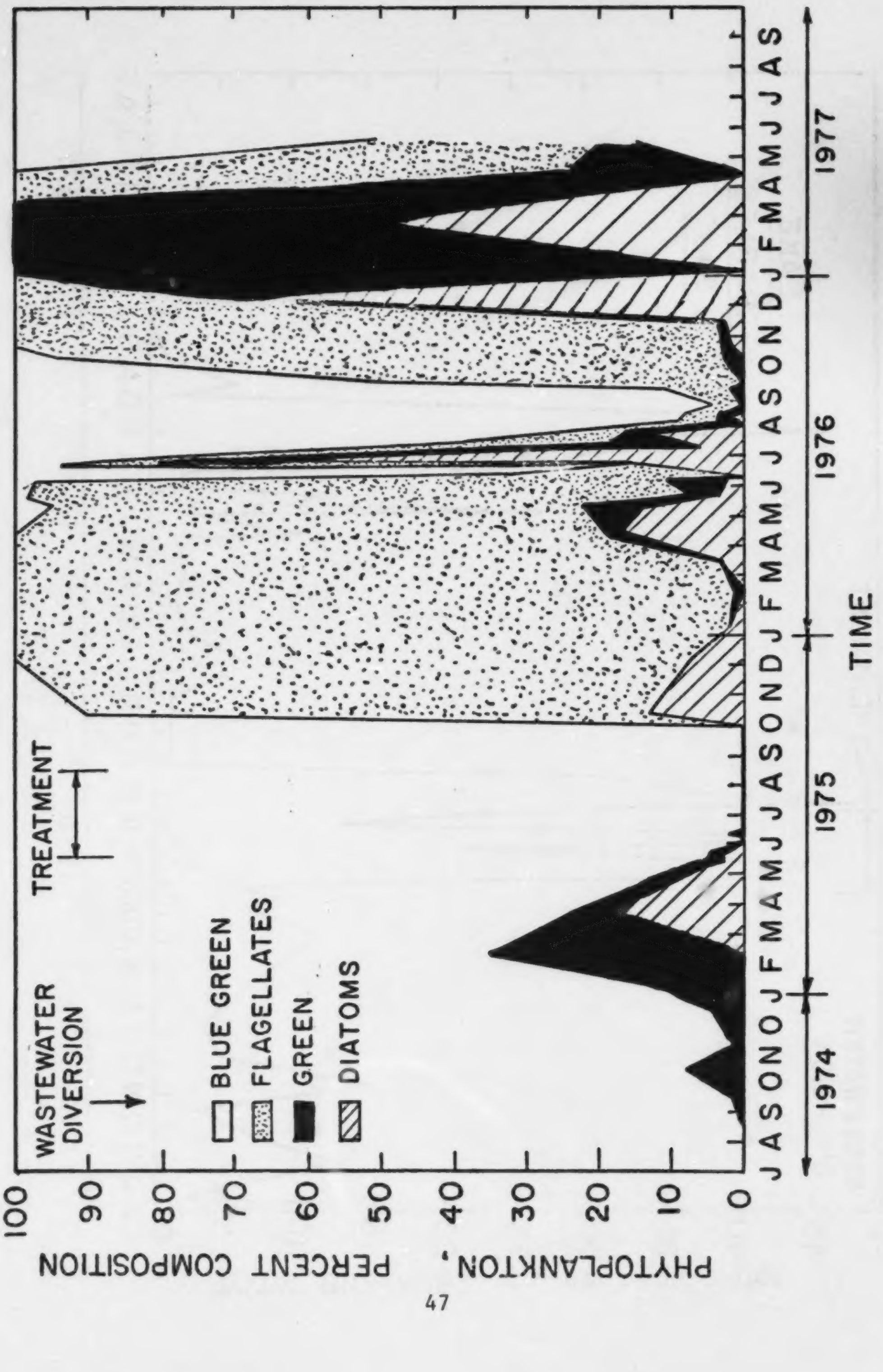

FIGURE 11: Phytoplankton Composition in Lake Charles East.

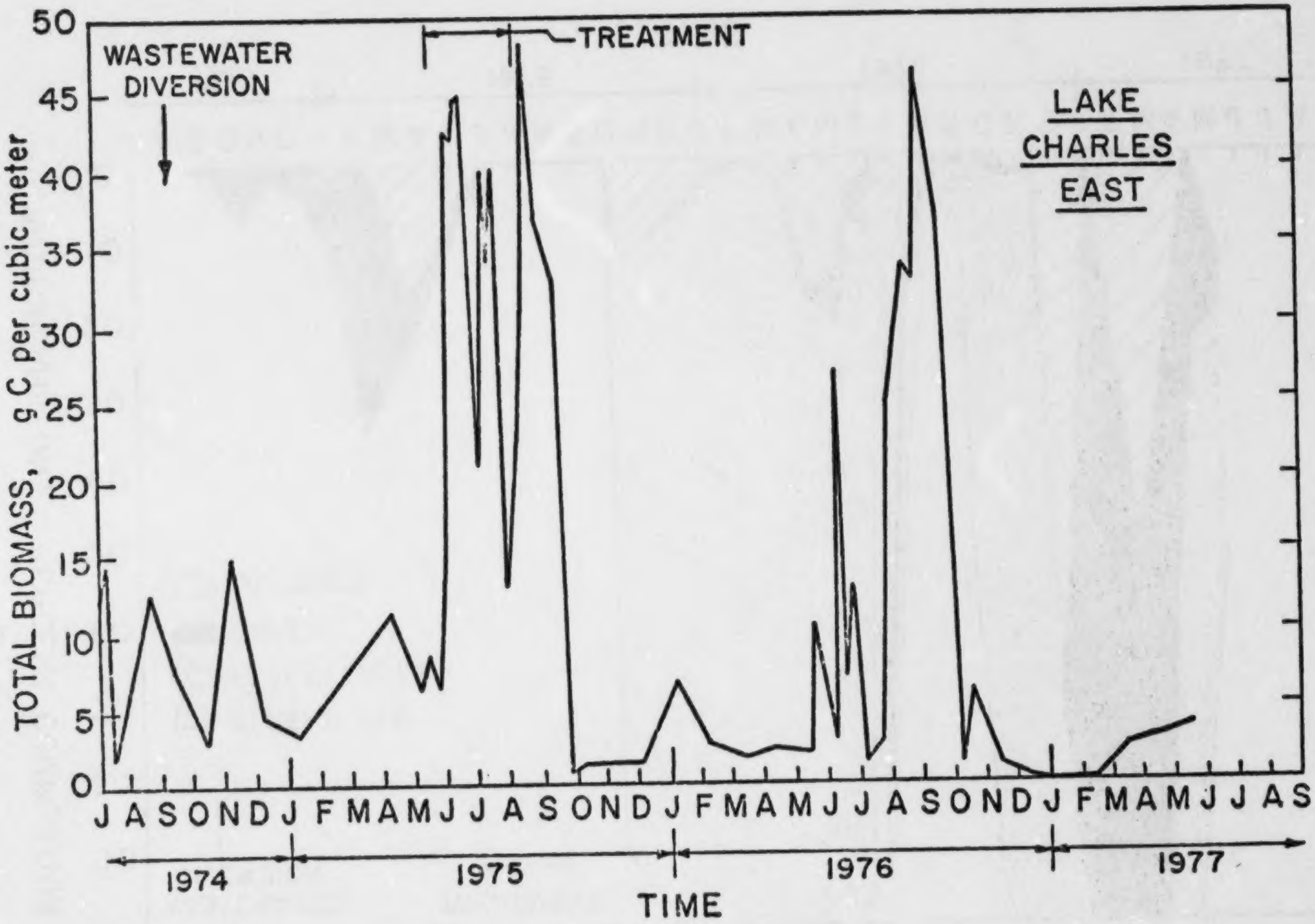

FIGURE 12: Phytoplankton Biomass Variation in Lake Charles East.

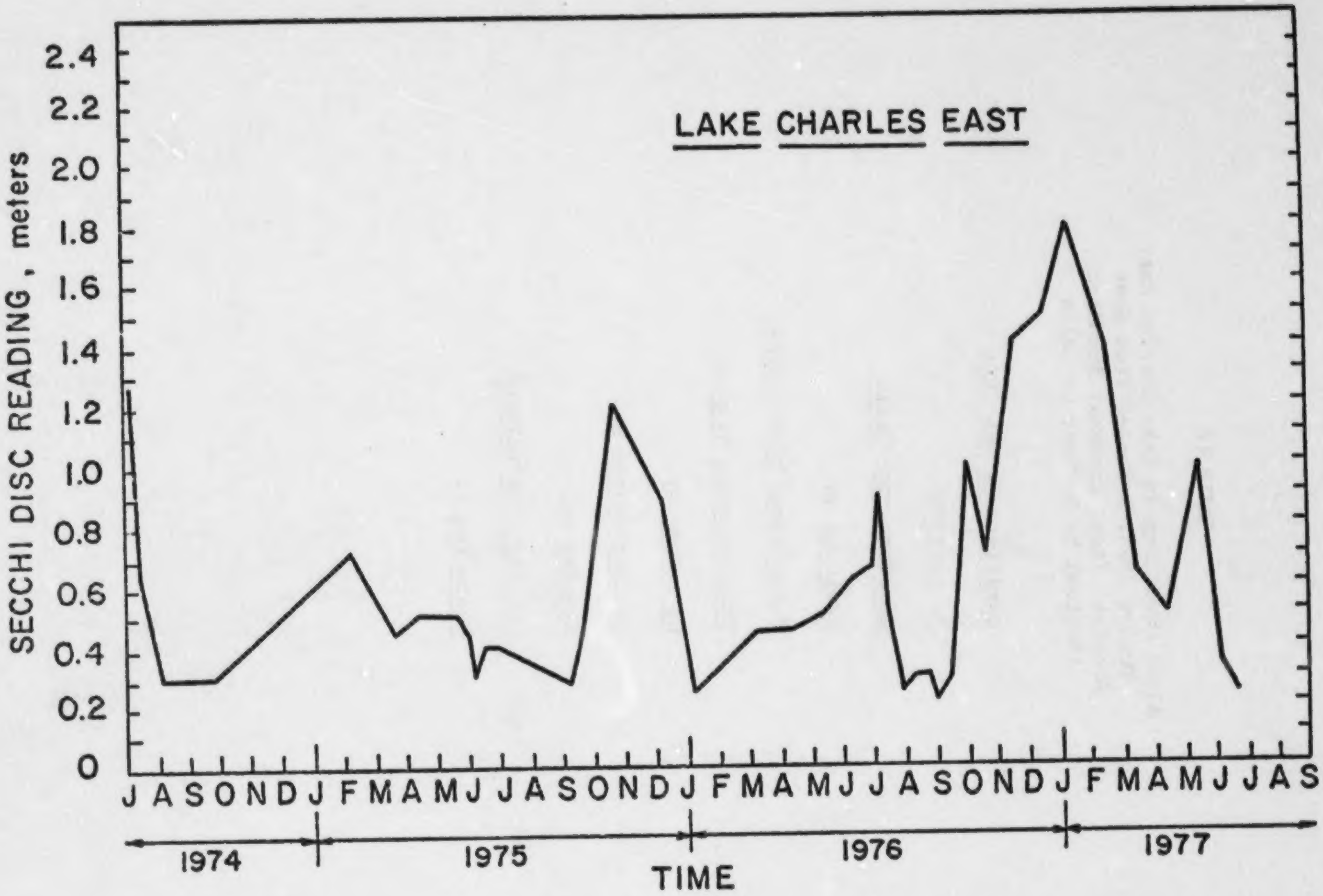

FIGURE 13: Secchi Disc Values for Lake Charles East.

TABLE 12

Algae Occurring in Lake Charles East During 1974, Arranged From Most Abundant (top) to Least Abundant (bottom) by Numbers of Cells

Oscillatoria agardhii

O. putrida

Anabaena circinalis

Navicula sp.

Scenedesmus quadricauda

Trachelomonas hispida

Gyrosigma sp.

Ankistrodesmus sp.

Synedra sp.

Oscillatoria vulgare

Cosmarium sp.

$g\text{-}C/m^3$ before treatment. Reference to Figure 11 indicates that the vast majority of this biomass was blue-green algae throughout the period from June 1974 to June 1975.

Figure 12 shows that a summer bloom of blue-green algae was being initiated just as the fly ash treatment began. Phytoplankton biomass climbed to relatively high levels, although fluctuations were evident. It should be noted, however, that a similar bloom occurred the following summer (1976). This point will be deferred to the Discussion of Results (Section VIII). The bloom observed during treatment (Summer of 1975) was dominated throughout by blue-green algae (Figure 11).

Shortly after the treatment of Lake Charles East with lime and fly ash, a significant change occurred in the phytoplankton composition. Figure 11 shows the appearance of flagellated species, dominated by Cryptomonas ovata, which by the end of 1975, had accounted for 98% of the phytoplankton biomass. By the time Cryptomonas became important in the lake, the total biomass of phytoplankton had dropped to rather low levels (Figure 12). The cryptophyte dominated in the lake through the winter.

In the Spring 1976, diatoms appeared and constituted about 15% of the total phytoplankton biomass in April. By the end of May the cryptophytes and diatoms had diminished and were briefly replaced by blue-green algae. Important species present at that time were Anabaena circinalis and Microcystis aeruginosa. Oscillatoria agardgii was also represented, but in much reduced numbers from previous years.

Coincident with the May bloom of blue-greens, was the appearance of Gymnodinium palustre, at one point reaching over 10% of the phytoplankton biomass. This dinoflagellate persisted in our samples until October. The blue-green bloom was also briefly interrupted by a bloom of diatoms, predominantly Cyclotella sp.

The blue-green algal bloom of 1976 was short-lived in comparison to those of the previous two years. The population was again replaced in October by the cryptophyte, Cryptomonas ovata. Diatoms were more common members of the phytoplankton community after treatment. Their presence was not greater in absolute numbers, but rather was evidenced by duration through the year.

In addition to the phytoplankton composition change (Figure 11), biomass levels changed as well. Figure 12 shows two changes following fly ash treatment. First, the summer bloom of 1976 showed less constancy over its duration than did the bloom of 1975. Indeed, biomass was considerably lower during the early Summer of 1976. It must be remembered, also, that the species composition was considerably different.

The other change is represented by reduced phytoplankton biomass in the period between the successive summer peaks. Again, the composition of the phytoplankton in relative percentage of biomass changed greatly during that time. Prior to treatment, blue-green algae dominated the year around, while after treatment, the cryptophyte became more important.

ZOOPLANKTON EFFECTS

The Rotifer Community

The composition of the zooplankton communities in Lake Charles East for the duration of the study are depicted graphically in Figure 14. Zooplankton trends are presented briefly below and are discussed in more detail in Section VIII. The following discussion is based on population counts (see Appendix).

The rotifer community in Lake Charles East exhibited two peaks each year. These two peaks were evident in 1975, 1976, and 1977 (first peak). The first population peak usually occurred in June and extended into July each year. It was then followed by a population low in August, with a second peak occurring between September and October. The second peak was less than half the magnitude of the first peak. Even though the species composition of each peak was different from year to year, the relative density at each peak was very similar. Spring peaks of all the years lie between 17,000 and 20,000 rotifers per liter of water sample, and the Fall sample ranged from 5,000 to 7,000 individuals per liter. According to the trophic classification system of Ruttner-Kolisko[35], the extremely high densities of rotifers found in Lake Charles East would place the lake in the eutrophic category for all the years studied. This trophic designation is further confirmed by the presence of the rotifers _Anuraeopsis fissa_, _Keratella cochlearis_ and _Pompholyx sulcata_, all of which are known to be good biological indicators of eutrophic lakes in North America[36].

Because of the relatively long time taken to apply the fly ash to the lake, the treatment itself did not seem to have any significant effect on the rotifer density. However, the rapid addition of lime at the end of the treatment process eliminated the rotifer community from the lake for somewhat less than a week. Repopulation of the treated side of the lake was rapid. The lime addition coincided with the seasonal low of rotifers in August, and therefore, the detrimental effect of lime was reduced in magnitude.

The Cladoceran Community

The effect of treatment on cladocerans in Lake Charles East followed the pattern set by the rotifers discussed above. That is, the lime application halted the development of cladoceran population in the lake, and numbers were not regained until several weeks later. The high density of *Bosmina* observed in the summer months of 1974 was not repeated in 1975 after the treatment, and in fact, never showed up again for the duration of the sampling program. Two species of *Daphnia* (*D. longiremis* and *D. ambigua*) were also present in the late summer and fall prior to treatment of the lake. *Alona* and *Ceridaphnia* were represented in those early collections as well.

The Copepod Community

Population of cyclopoid copepods (*Cyclops* and related genera) showed considerable fluctuation during the late summer and fall of 1974, through early 1975. The annual trend for total copepods is similar to that of the cladocerans in Lake Charles East, in that both have spring and fall population peaks separated by a summer minimum. The spring peaks, however, were dominated by nauplius larvae, with adults constituting only a minor component of the population. In the fall, population peaks of nauplii were followed by high densities of adult copepods. Immediately prior to the application of lime and fly ash, the spring peak was already in its decline.

The short-term effect of the chemical treatment seems to be the termination of the later part of the annual cycle of the copepods by eliminating the aestivating copepodites in the summer (see Section VIII). If the treatment

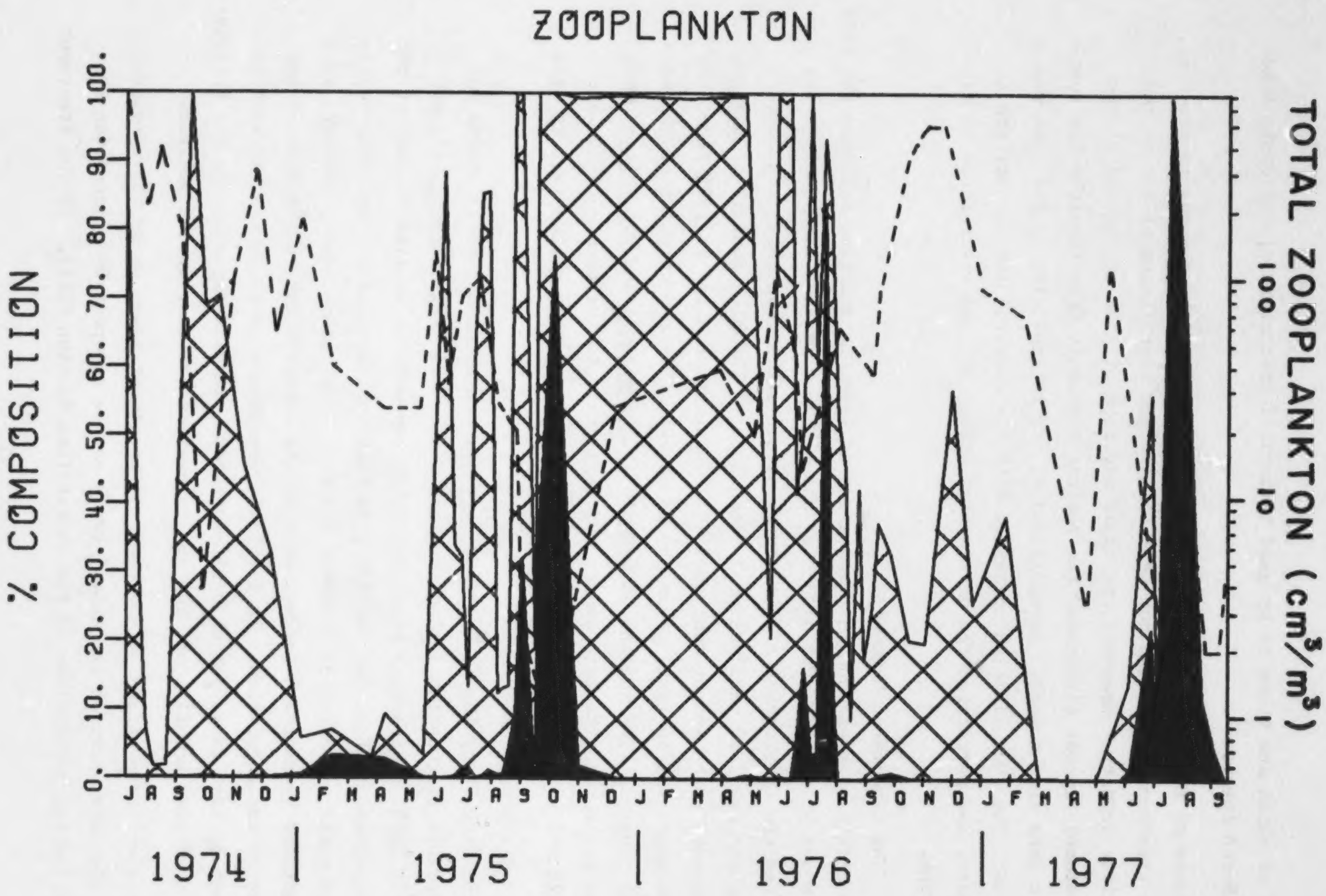

Figure 14: Zooplankton Composition in Lake Charles East Solid = Rotifers, Crossed = Cladocera, White = Copepods, Dotted Line = Total Zooplankton.

were applied at another time, the copepod community may be affected less significantly when the individuals are planktonic and do not have to contend with the fly ash sediments.

Fish

The major fish species in Lake Charles East are the bluegill (*Lepomis macrochirus*) and the largemouth bass (*Micropterus salmoides*). While no systematic study was carried out, a few observations can be made. A gill net set in Summer 1974, showed the lake to be populated primarily with stunted bluegill. In the Fall of 1974, a fish-kill occurred which substantiated the gill-net observations. The kill was attributed to circulation of the lake (resulting in reduced oxygen levels) coupled with elevated ammonia levels (determined by personnel from the Steuben Co. Health Department). Virtually all bluegill that died were in the 10-12 cm range. Some large (30 cm) bass were also killed.

A fish kill occurred in the Summer of 1975, as a result of the liming operation associated with the lake treatment. Several thousand small bluegill died on the side of the lake being treated. No dead bass were observed (see Section VIII).

Benthos

The benthic community of Lake Charles East both before and after treatment consisted primarily of chironomid larvae, a common indicator of a eutrophic status. The effects of the treatment on this and other sediment-dwelling creatures appeared to be minimal. Data for surveys performed the year before (1974) and the year after treatment (1976) are given in Table 13. Chironomids and *Chaoborus* were more abundant after treatment while fewer oligochaetes were found.

SEDIMENT CHEMISTRY EFFECTS

Inasmuch as the main function of the added fly ash was to bring about a retardation of phosphate release from the sediments of Lake Charles East, a major portion of this study was devoted to an evaluation of the effectiveness of the ash seal. It has previously been indicated in Section VI the extent of coverage of the treated sediments (Figure 6). Treatment of only

TABLE 13

Lake Charles East Benthos Data
(individuals/ft^2)

12-6-74 (before treatment)

Depth (m)	Oligochaeta	Chironomus plumosus	Palpomyia tibialis	Chaoborus punctipennis
2.4	129	67	6	0
1.8	45	110	27	0
1.8	101	303	39	0
1.4	50	45	18	0
2.4	0	20	2	7
2.6	3	44	4	4
2.9	0	4	1	3
2.7	1	139	2	9
2.7	28	92	6	0

2-20-76 (after treatment)

Depth	Oligochaeta	Chironomus	Unknown	Chaoborus
0.8	84	460	0	0
1.5	4	288	0	0
2.1	4	364	16	4
2.5	0	184	0	4
2.8	0	44	0	48
2.8	0	4	0	24
3.1	4	4	0	480
3.0	0	0	0	84
2.9	0	12	0	64
2.8	0	44	0	52
2.5	12	128	0	20
2.0	0	348	0	0

a portion of the lake provided control sediments (western portion of the lake) to which comparisons could be made. Parallel sets of undisturbed sediment cores were taken from both sides of the lake (WILDCO Model 2404 K.B. core sampler) on a monthly basis during 1976, the year following treatment. Variations in pertinent lake parameters during this time are summarized in Table 14. For each set of cores gathered, a complete fractionation of sediment-bound phosphorus into the citrate-dithionite-bicarbonate (CDB) extractable, apatite, remaining non-apatite forms, and organically bound phosphorus was made at several sediment depths. The procedures are summarized in Figure 15. Soluble interstitial phosphorus was also determined. Other important sediment parameters determined included CDB-Fe, pH, moisture content, and specific gravity of sediment solids. Iron analyses were run by flame atomic absorption.

Remaining cores were incubated in the dark in a batch mode under prevailing conditions of dissolved oxygen and temperature in a sealed environmental control chamber (Aminco Model 4-5460) for measurement of release and sorption of phosphorus. Figure 16 shows a schematic diagram of the apparatus. A small amount of stirring was provided in order to give a more completely mixed system above the sediment-water interface. Phosphate analyses on the overlying water were made daily. Certain aerobic cores which had a comparatively low phosphorus level above the sediments were spiked with small amounts of phosphate to provide greater accuracy in measurement differentials during sorption experiments.

Concern over the potential adverse environmental effects of heavy metals associated with fly ash led to the analysis of cores from both sides of the lake for these constituents. The top five centimeters of the cores were dried (103°C) and digested via a hot concentrated nitric acid reflux method. Other cores were centrifuged to bring about a good separation of solids from interstitial water. The supernatant was filtered (0.45 μm) and analyzed for soluble metal concentrations. All metal analyses in this phase of the study were made by flameless atomic absorption (Perkin-Elmer Model 305 with HGA-2100 graphite furnace).

Figures 17 and 18 show total residue and volatile residue, respectively, of

TABLE 14

Seasonal Variation of Selected Limnological Parameters
for Lake Charles East, 1976

Parameter	4/28	6/16	7/14	8/18	9/15	10/20	11/17	12/10
Secchi Disc (m)	0.45	0.62	0.55	0.37	0.30	0.70	1.4	2.4
Surface Temp (°C)	13	29	-	23	21	9	4.5	3
Bottom Temp (°C)	12	23	-	21	19	9	3	3
Soluble Phosphorus Bottom (μg-P/ℓ)	170	130	400	60	50	100	90	80
Oxygen Conditions	Aerobic	Anaerobic	Anaerobic	Anaerobic	Anaerobic	Aerobic	Aerobic	Aerobic

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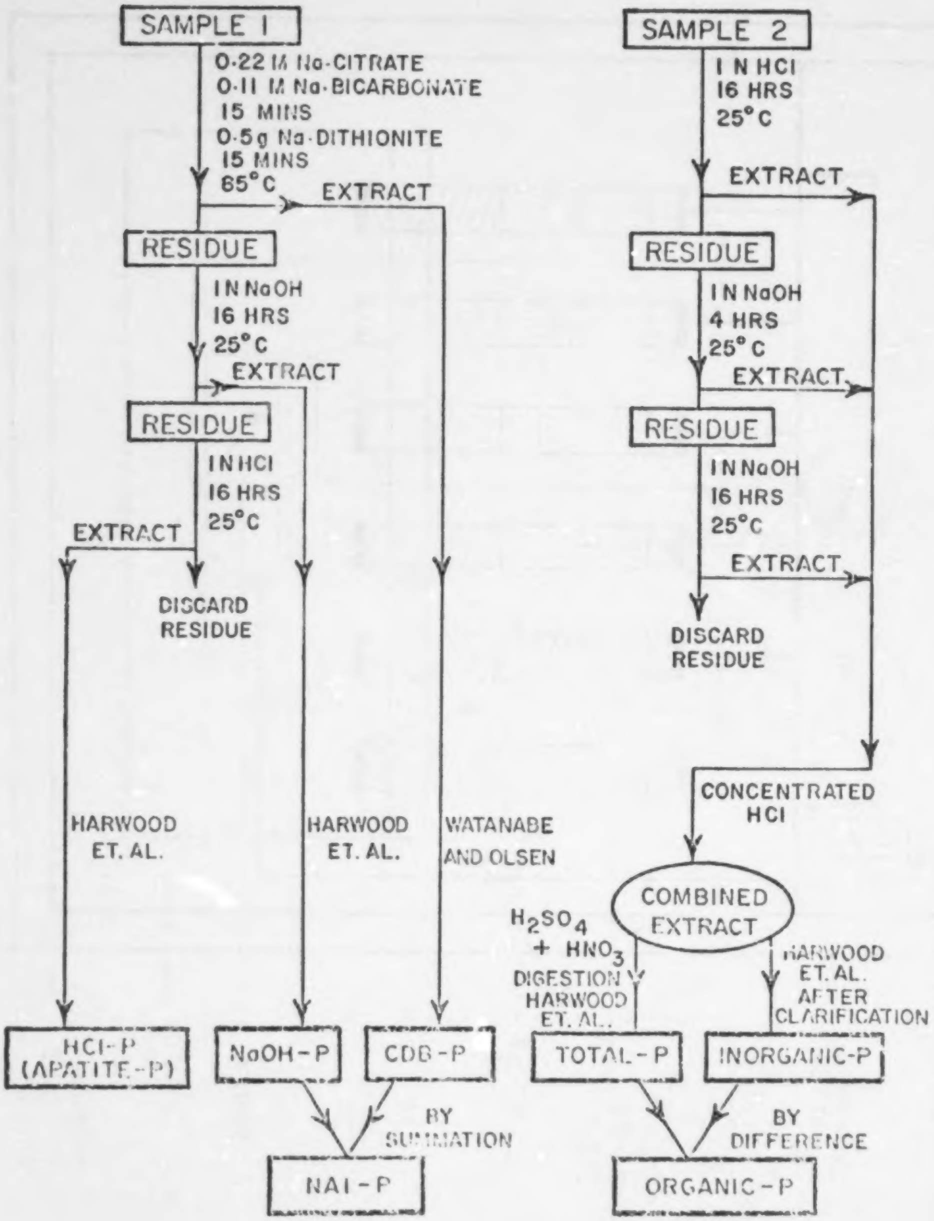
SAMPLE 1
0.22 M Na-CITRATE
0.11 M Na-BICARBONATE
15 MINS
0.5 g Na-DITHIONITE
15 MINS
85°C
EXTRACT
RESIDUE
1 N NaOH
16 HRS
25°C
EXTRACT
RESIDUE
1 N HCl
16 HRS
25°C
EXTRACT
DISCARD RESIDUE
HARWOOD ET. AL.
HARWOOD ET. AL.
WATANABE AND OLSEN
HCl-P (APATITE-P)
NaOH-P
CDB-P
BY SUMMATION
NAI-P
SAMPLE 2
1 N HCl
16 HRS
25°C
EXTRACT
RESIDUE
1 N NaOH
4 HRS
25°C
EXTRACT
RESIDUE
1 N NaOH
16 HRS
25°C
EXTRACT
DISCARD RESIDUE
CONCENTRATED HCl
COMBINED EXTRACT
H2SO4 + HNO3 DIGESTION HARWOOD ET. AL.
HARWOOD ET. AL. AFTER CLARIFICATION
TOTAL-P
INORGANIC-P
BY DIFFERENCE
ORGANIC-P

FIGURE 15: Phosphorus Fractionation and Analysis Scheme Followed for Lake Charles East Sediments.

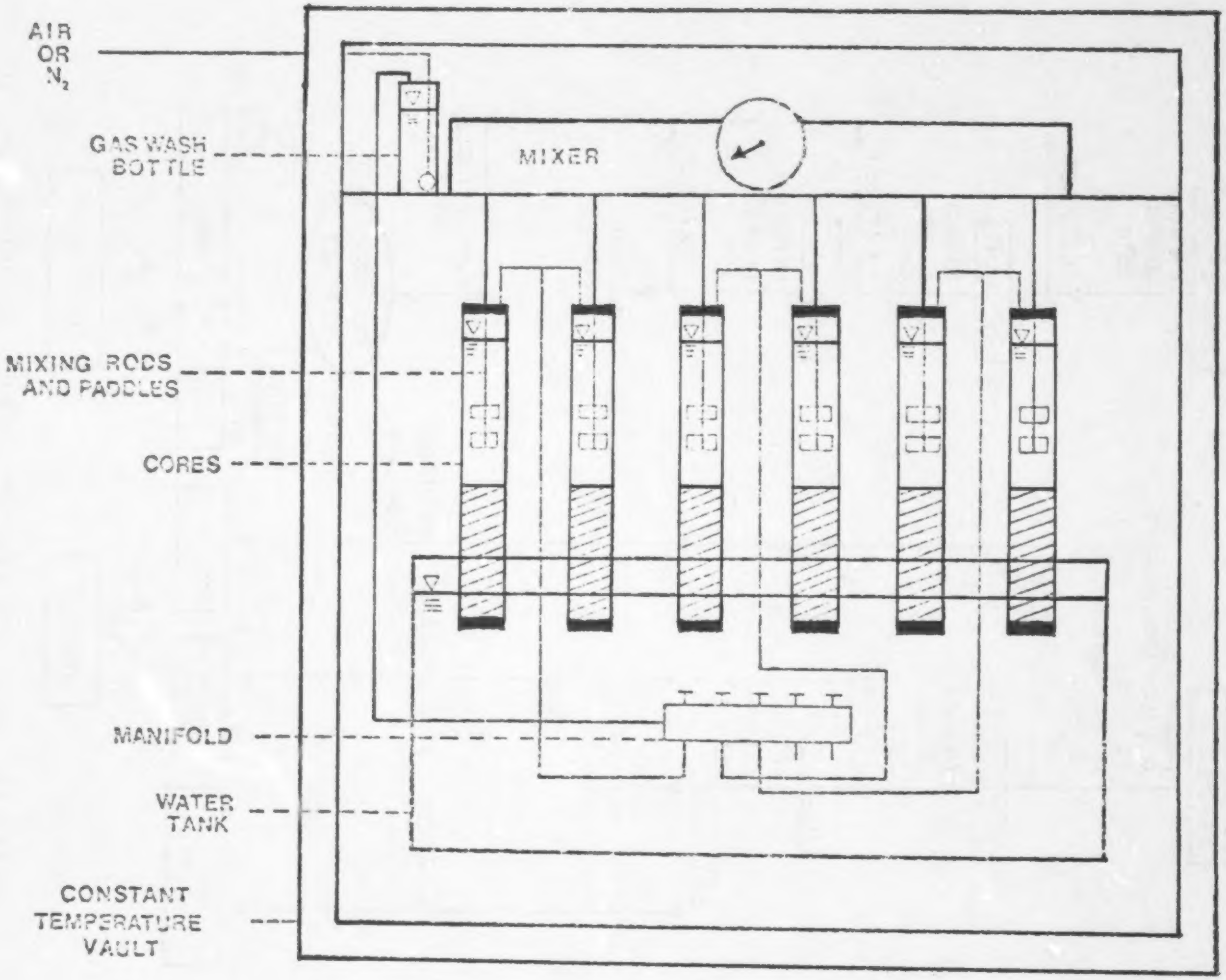
AIR OR N_2
GAS WASH BOTTLE
MIXER
MIXING RODS AND PADDLES
CORES
MANIFOLD
WATER TANK
CONSTANT TEMPERATURE VAULT

FIGURE 16: Schematic of Core Incubator.

sediment material as a function of depth for treated and untreated lake sediments. The fly ash layer (about 5 cm) is clearly evident due to its much lower moisture content and volatility. The zone of higher residue but approximately constant volatility at 10-12 cm in the control sediments (12-15 cm on the treated side) may be due to the construction activity associated with the placement of the bridge foundation in 1967. If so, the rate of sedimentation in Lake Charles over this ten-year period is approximately 1 cm/yr, a high value but not out of line with lakes of such high productivity. In connection with this it was observed visually, and is evident in Figures 17 and 18, that a one to two centimeter layer of organic detrital material existed above the fly ash layer when samples were taken (about one year after treatment).

Some of this layer could have come about through initial mixing of sediments during treatment with a subsequent settlement on the fly ash. In view of the probable rate of sedimentation in the lake it appears more likely that most of the material was produced autochthonously after treatment. This phenomenon is important since it impacts on the degree of effectiveness of the seal and the permanence of the overall ash treatment. Implications will be discussed in Section VIII.

The seasonal variation in phosphorus release and uptake for both control and treated sediments for the period of study is given in Figures 19 and 20. Both sediments displayed uptake during aerobic periods and release when the lake bottom was anaerobic (refer to Table 14). Treated sediments, however, released considerably less phosphorus during the summer. When all anaerobic data are pooled the control side showed 31 ± 39 $mg/m^2/day$ for release of phosphorus while the treated side gave 14 ± 12 $mg/m^2/day$. The means were significantly different at the 95% confidence level. Uptake during aerobic periods was more nearly the same for both sides, 16 ± 12 $mg/m^2/day$ for the control and 13 ± 8 $mg/m^2/day$ for the treated sediments.

A comparison of release rates by months from June through September is shown in Table 15. Here it can be seen that the month of maximum P release for untreated sediments is July. The reduction in P release brought about by sediment treatment is significant at the 99% confidence level. This is

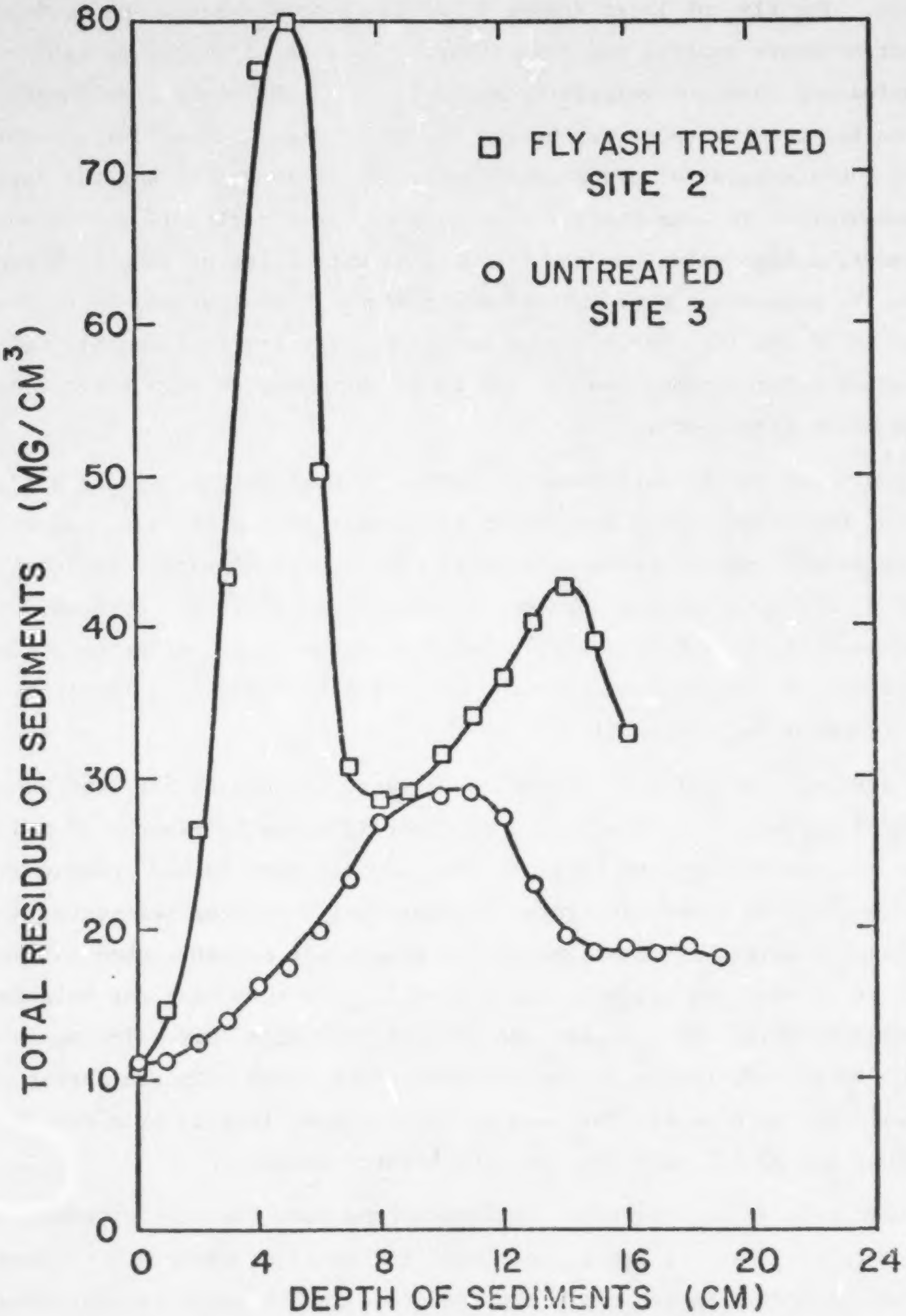

FIGURE 17: Density of Sediments in Lake Charles East vs. Sediment Depth

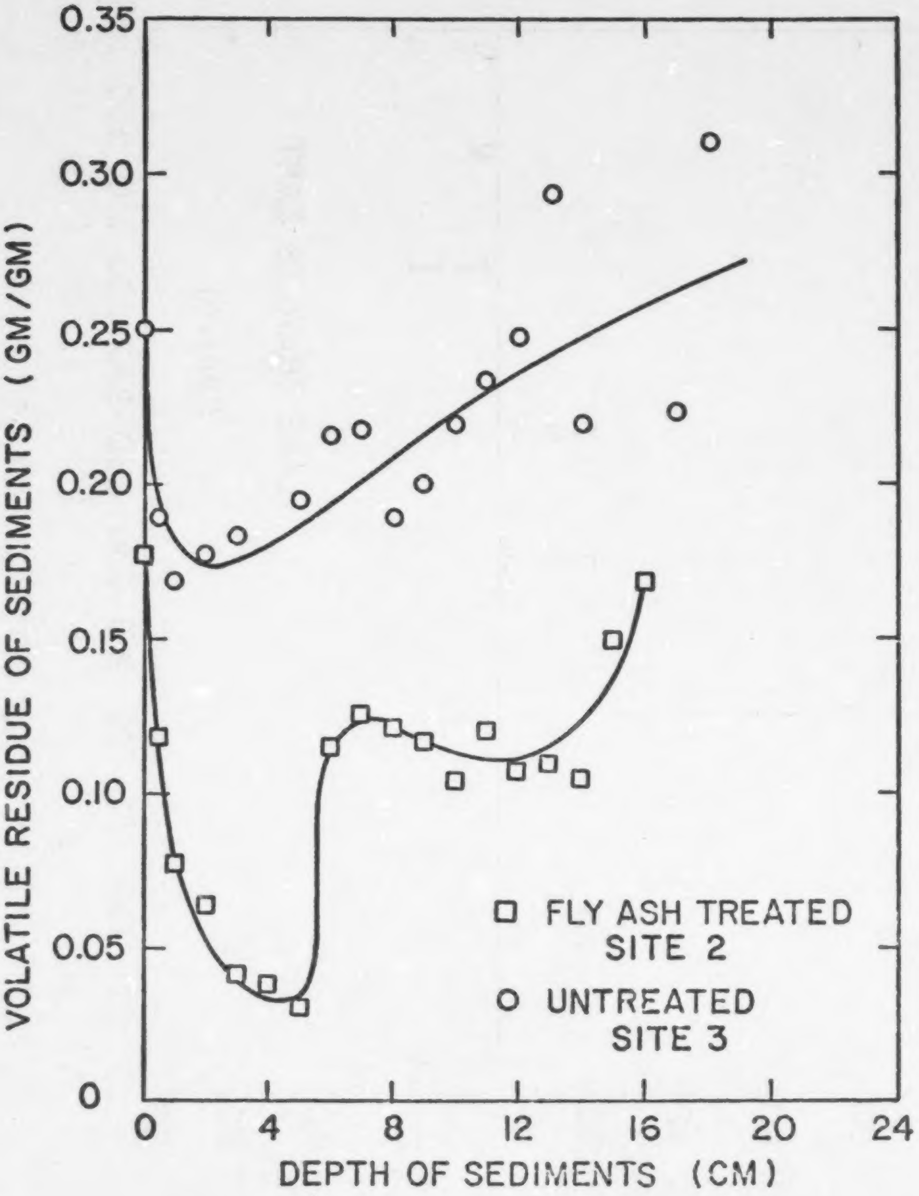

FIGURE 18: Fractional Volatile Residue in Sediments of Lake Charles East vs. Sediment Depth.

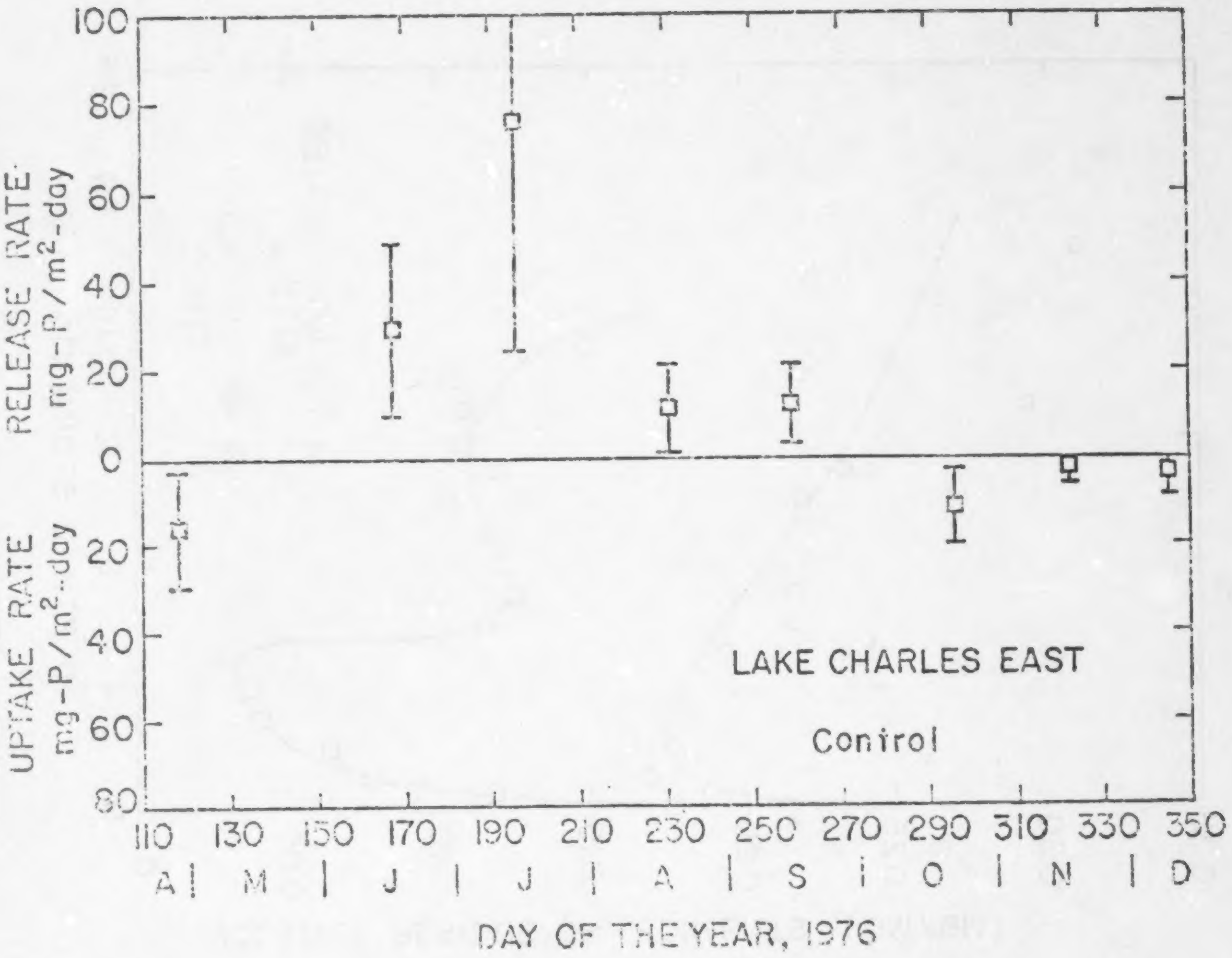

Figure 19: Phosphorus Release and Uptake in Lake Charles East Untreated Sediments.

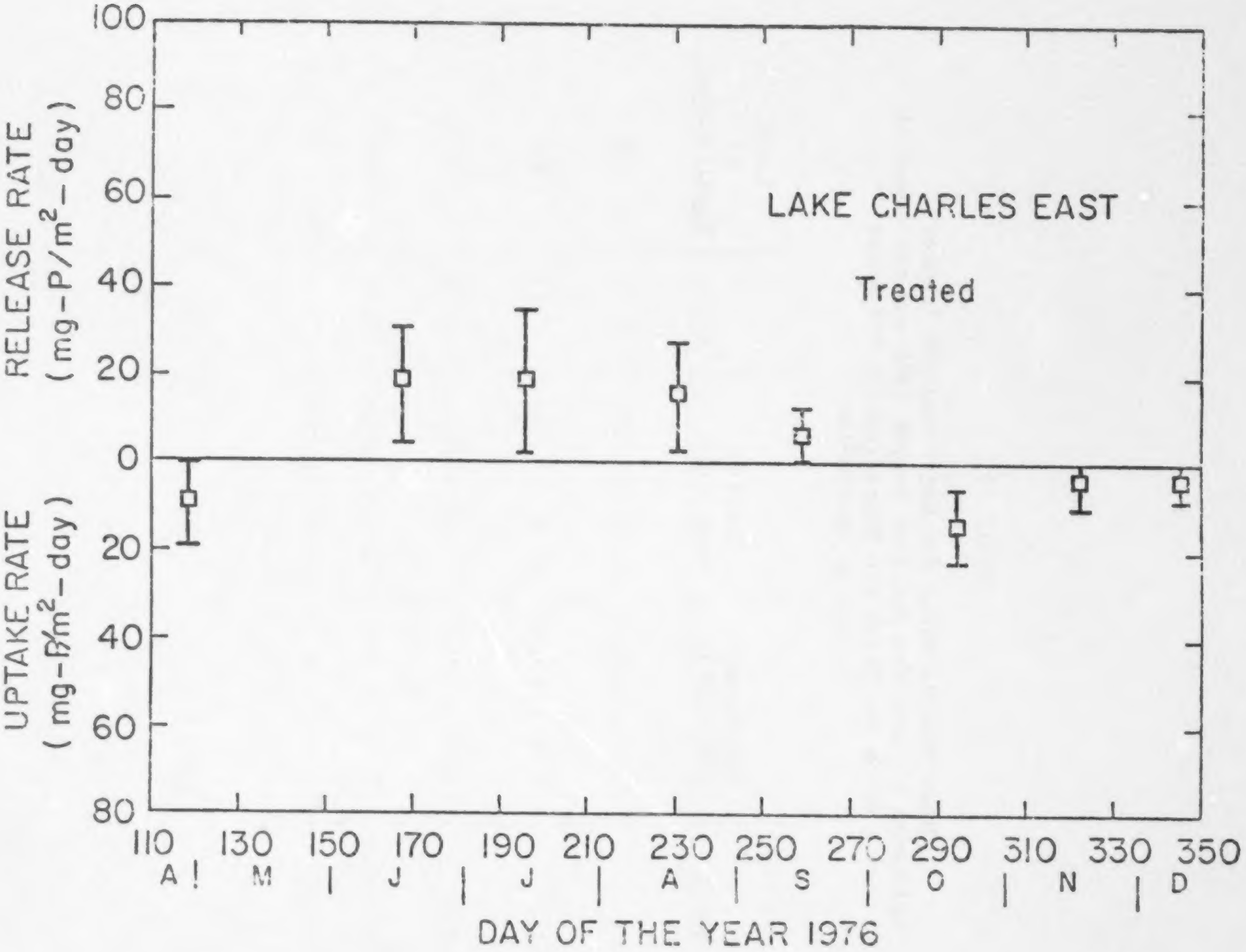

Figure 20: Phosphorus Release and Uptake in Lake Charles East Treated Sediments.

TABLE 15

Mean Release Rates for Untreated and Treated
Sediments of Lake Charles East During 1976 and the Level of
Which the Means are Significantly Different
(units of mg-P/m^2/day)

Month	Untreated Mean (STD)	Treated Mean (STD)	t_{score}	Level of Significance
June	22.27 (15.84)	17.55 (13.25)	0.75	50%
July	75.49 (48.42)	18.38 (14.54)	3.72	99%
August	11.18 (9.05)	15.42 (10.58)	-1.11	60%
September	11.19 (8.24)	6.75 (5.71)	1.47	80%

important since it shows that while fly ash treated sediments do release phosphorus under reducing conditions, the peak releases are dampened considerably.

The variation in important sediment phosphorus parameters are given in Figures 21 and 22 for the control and treated sediments, respectively. In Figure 22, for treated sediments, a distinction is made between the top centimeter or so of detrital material indicated previously and the ash layer. Total depth is to approximately six centimeters. Figure 21 gives readings according to depth for the top centimeter and every two centimeters to a depth of seven. A comparison of these figures indicates clearly the lower amounts of different phosphorus forms in the sediments treated with fly ash. Of special note are Figures 21 (b) and 22 (b), CDB extractable phosphorus, which is a measure of the most labile forms. It represents, in essence, a "reservoir" of phosphorus to the lake. The lower available amounts of phosphorus are reflected in Figures 21 (c) and 22 (c), soluble interstital phosphorus, which is also lower for the treated sediments.

In spite of these lower values for the treated sediments, the effect of the detrital layer above the ash is evident in Figure 22. It is reasonable to assume that the higher values of CDB and hence interstitial phosphorus in this layer are at least partially responsible for the release characteristics noted in Figure 20.

Further evidence for the importance of the citrate-dithionite-bicarbonate (CDB) extractable mineral forms is presented in Figures 23 and 24. Sediment values for CDB-iron values have been regressed on values for CDB-phosphorus yielding a positive correlation at the 99% level of significance. Although the slopes of the lines in Figures 23 and 24 are similar, the intercept for the treated sediments is much lower than the control reflecting the lower amounts of amorphous iron mineral forms in the treated sediments. Thus, the lower quantities of phosphorus which are available from the treated sediments (Figure 22 (b)) appear to be due to the smaller capacity of this fly ash to act as a phosphorus sink.

A summary of the effects of fly ash on the sediments of Lake Charles East is

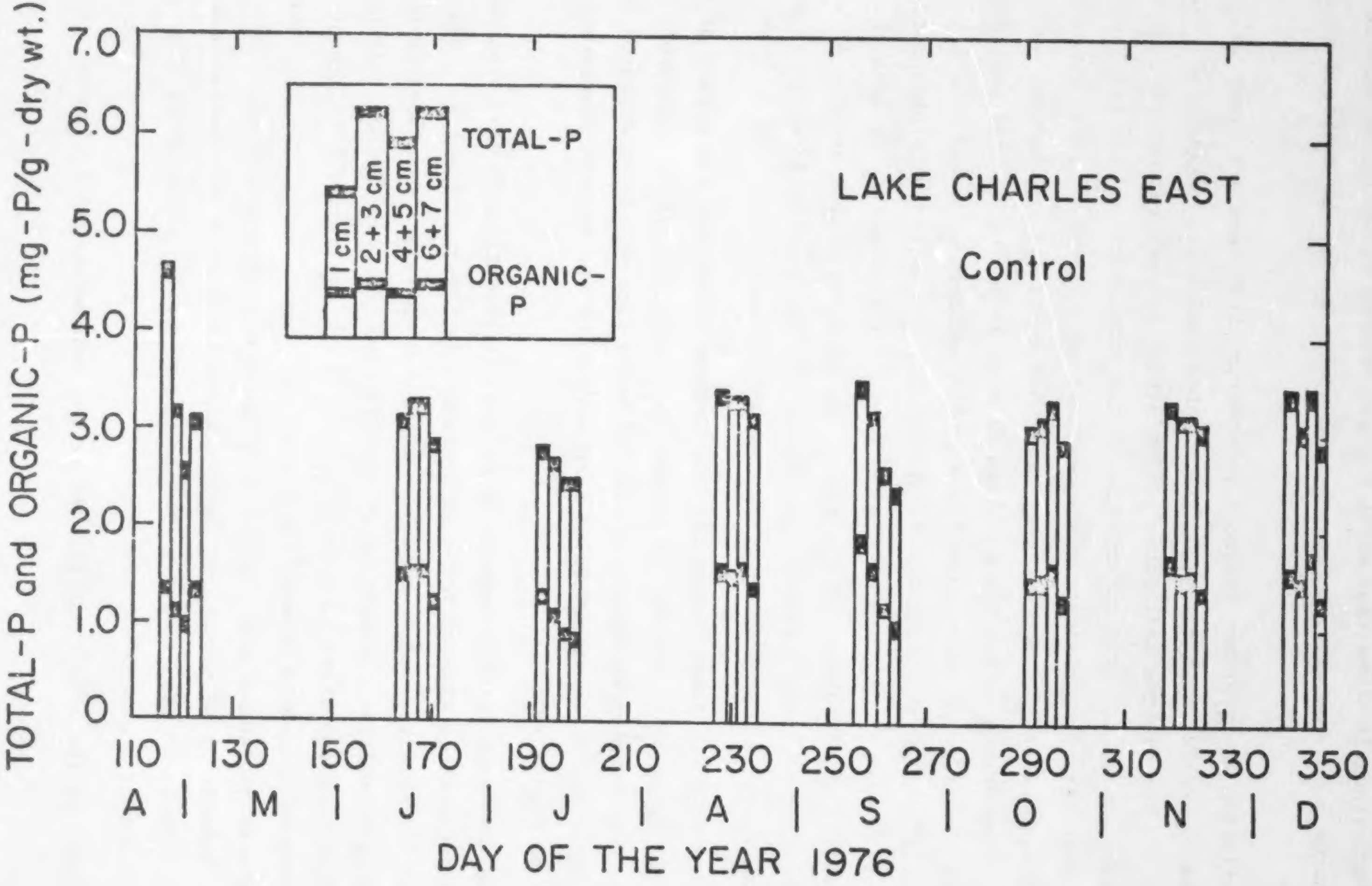

Figure 21(a): Total and Organic Phosphorus in Lake Charles East Control Sediments.

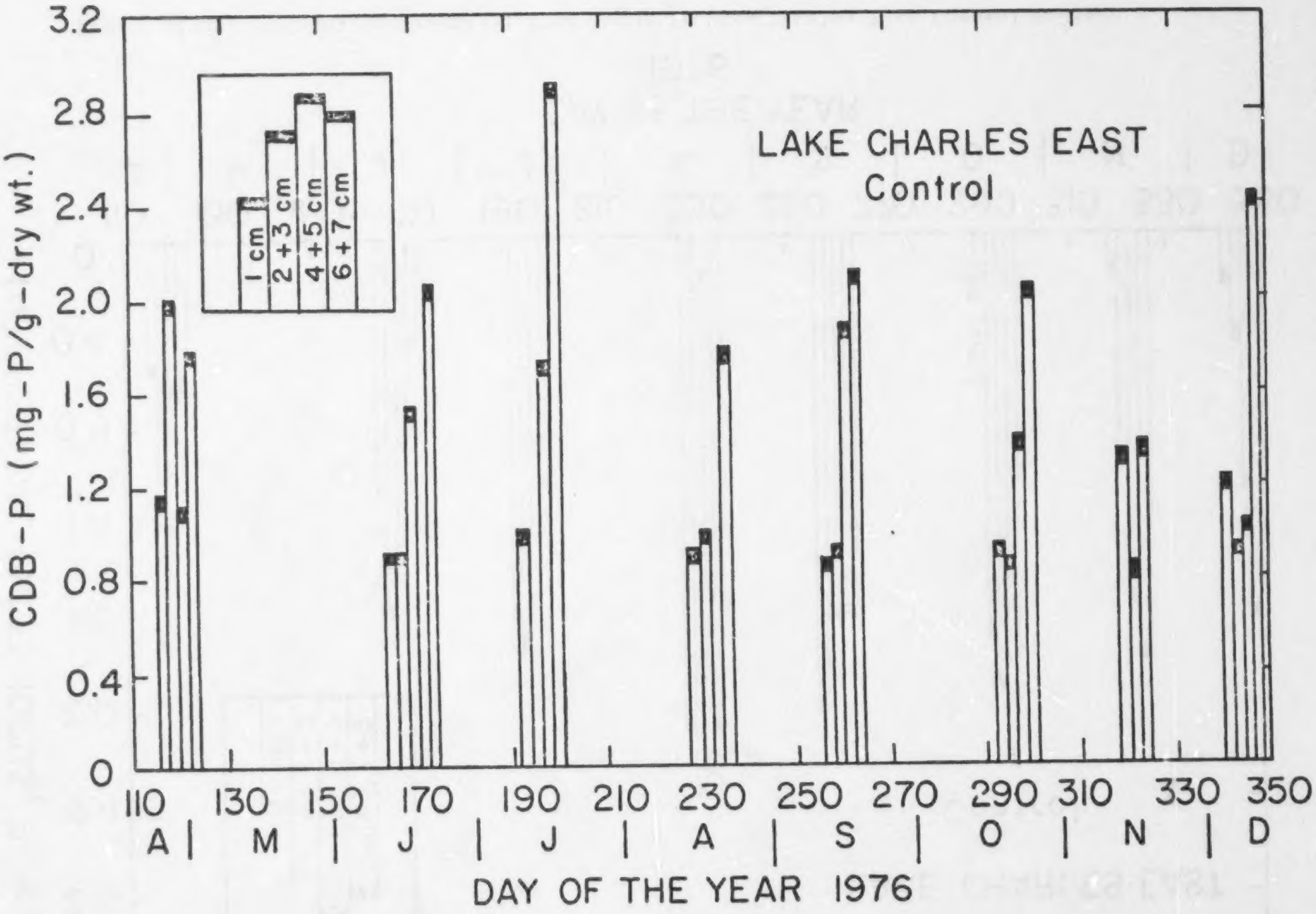

Figure 21(b): CDB Extractable Phosphorus in Lake Charles East Untreated Sediments.

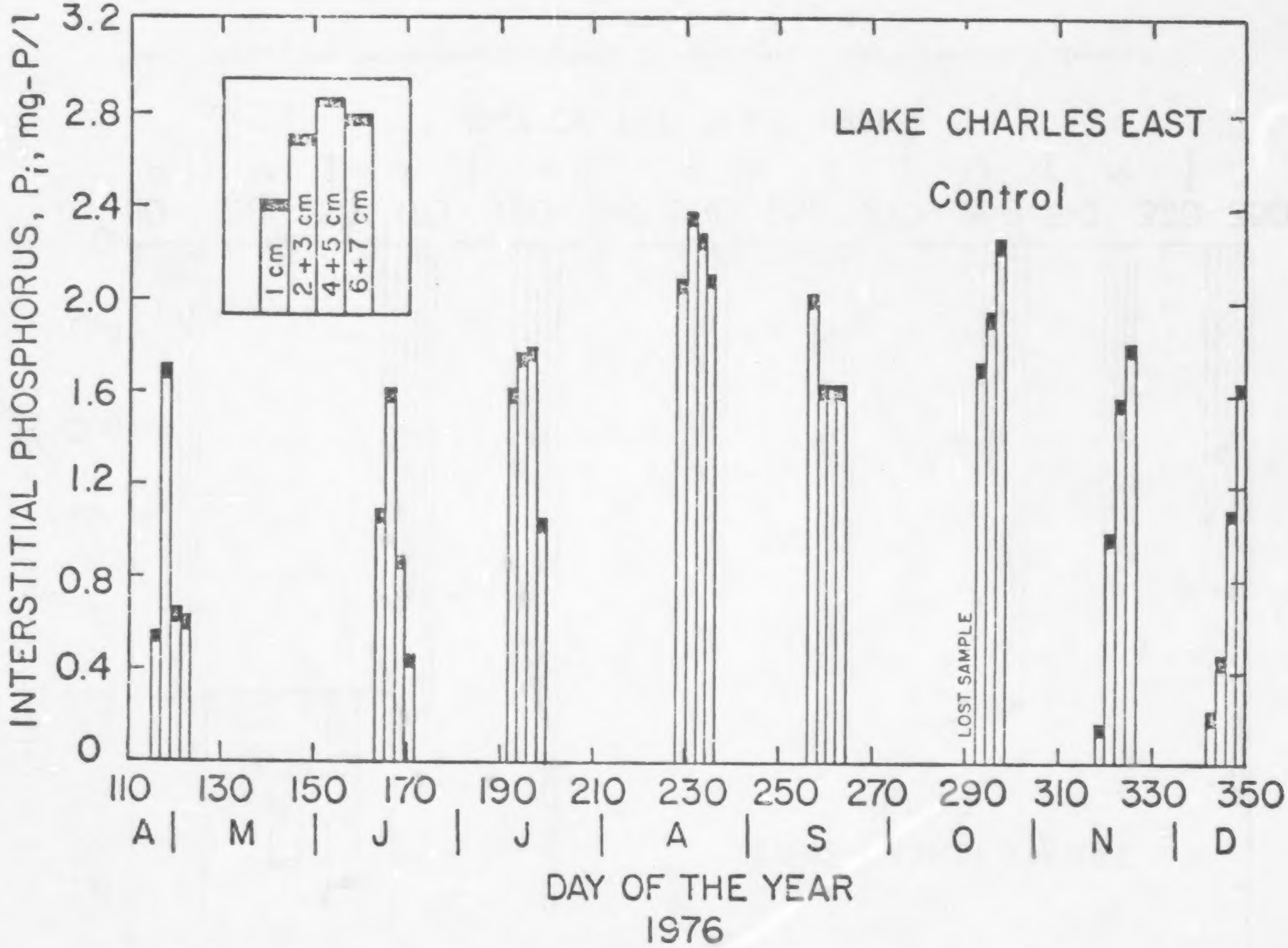

Figure 21(c): Soluble Interstitial Phosphorus in Lake Charles East Control Sediments.

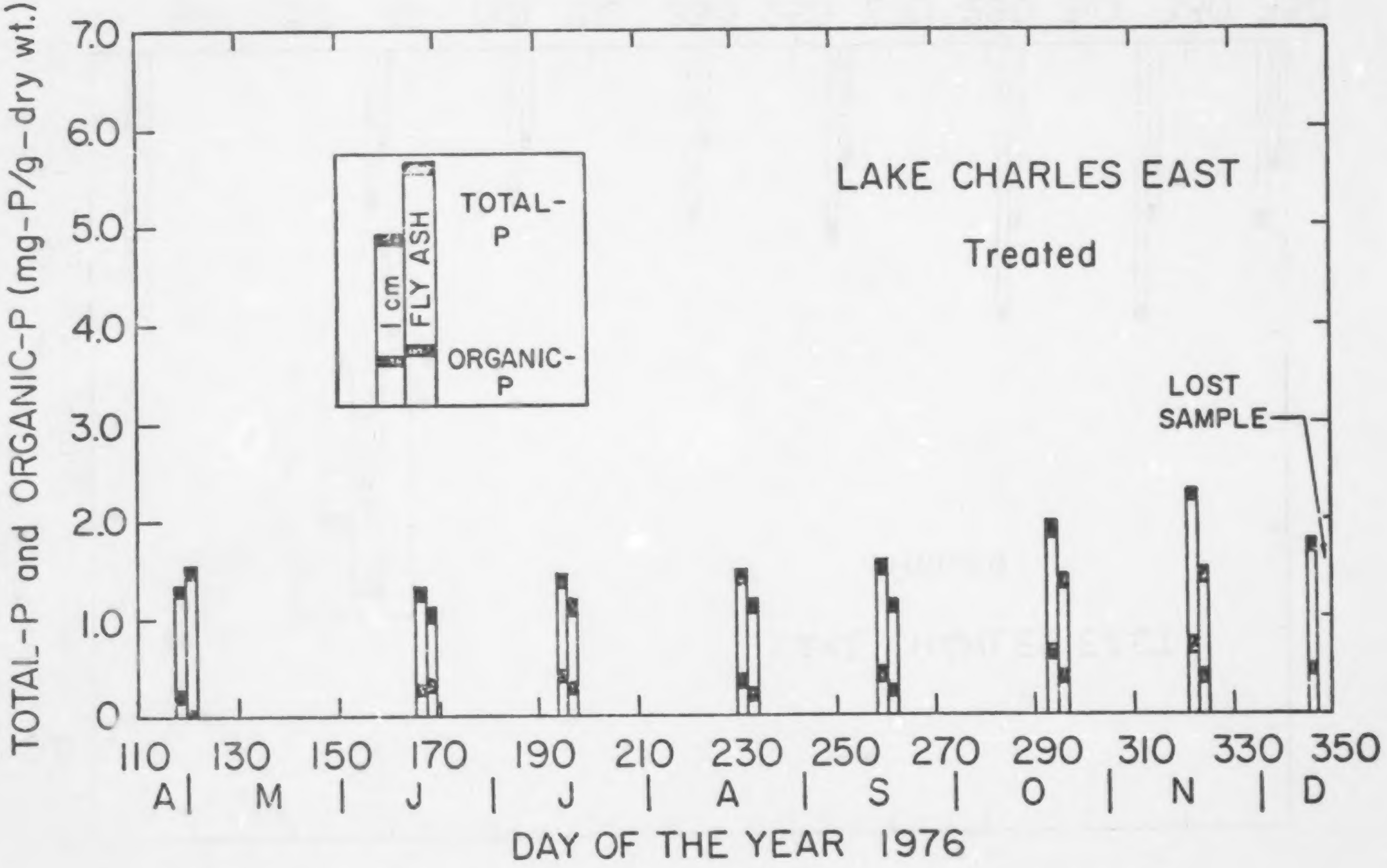

Figure 22(a): Total and Organic Phosphorus in Lake Charles East Treated Sediments.

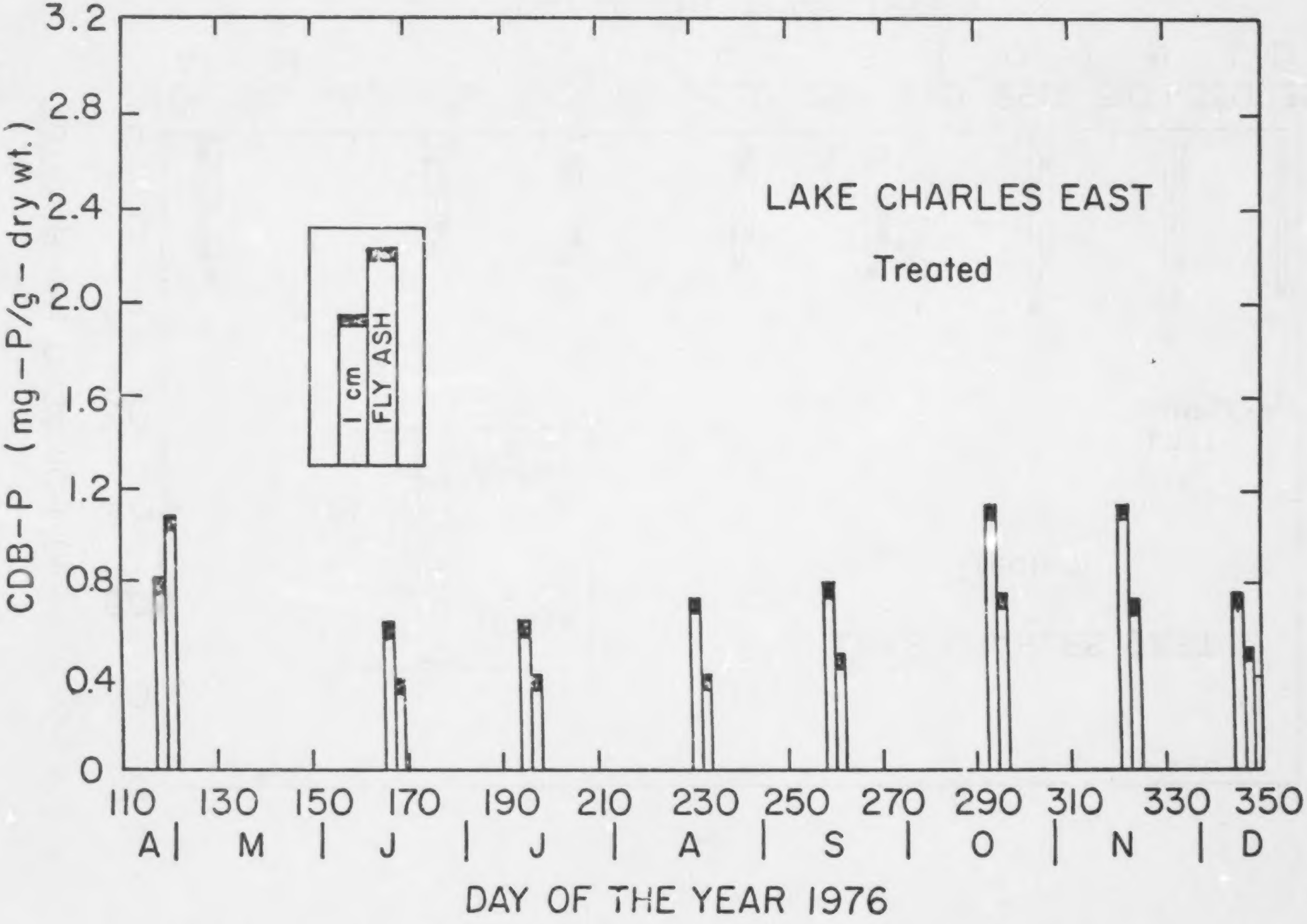

Figure 22(b): CDB Extractable Phosphorus in Lake Charles East Treated Sediments.

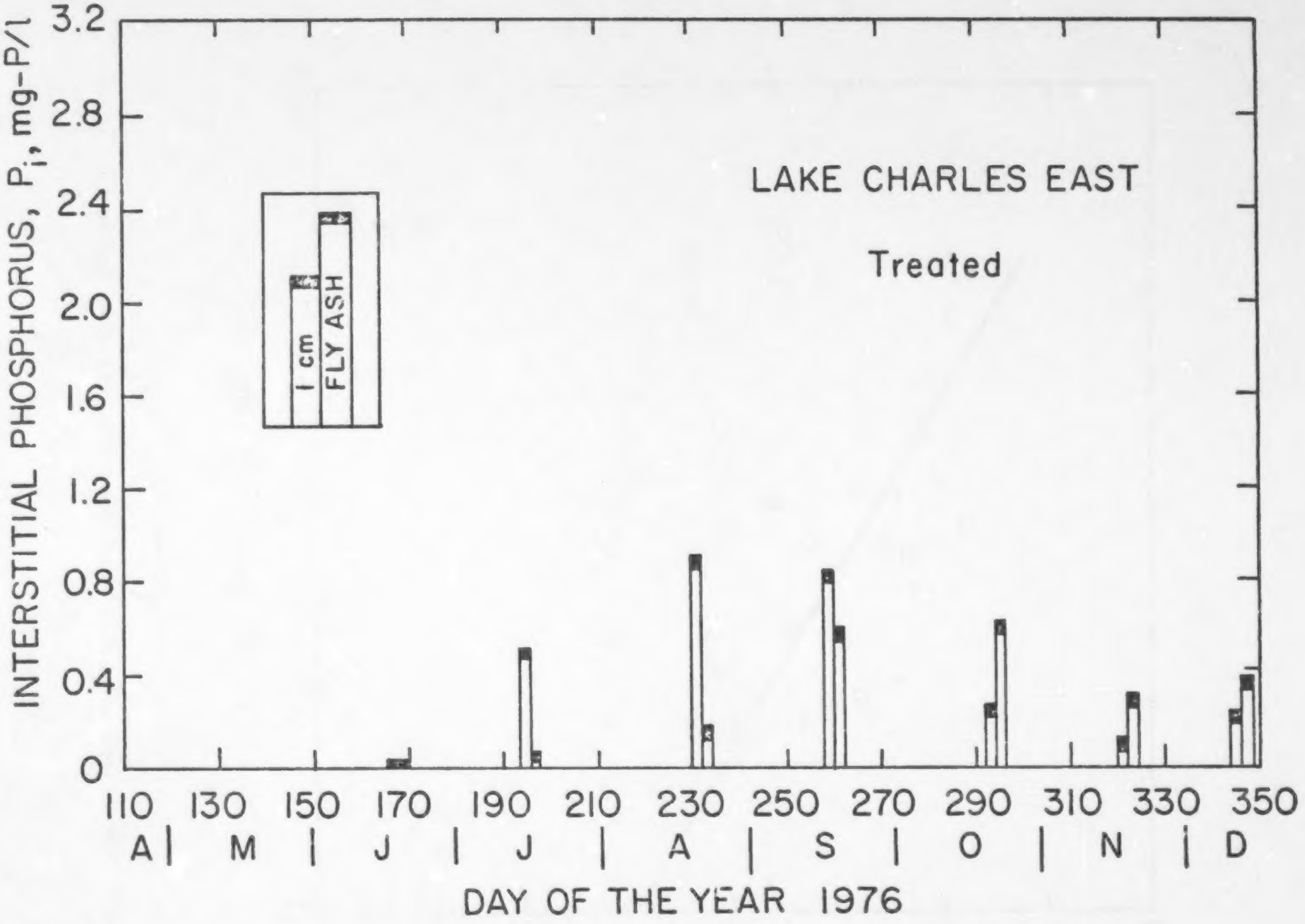

Figure 22(c): Soluble Interstitial Phosphorus in Lake Charles East Treated Sediments.

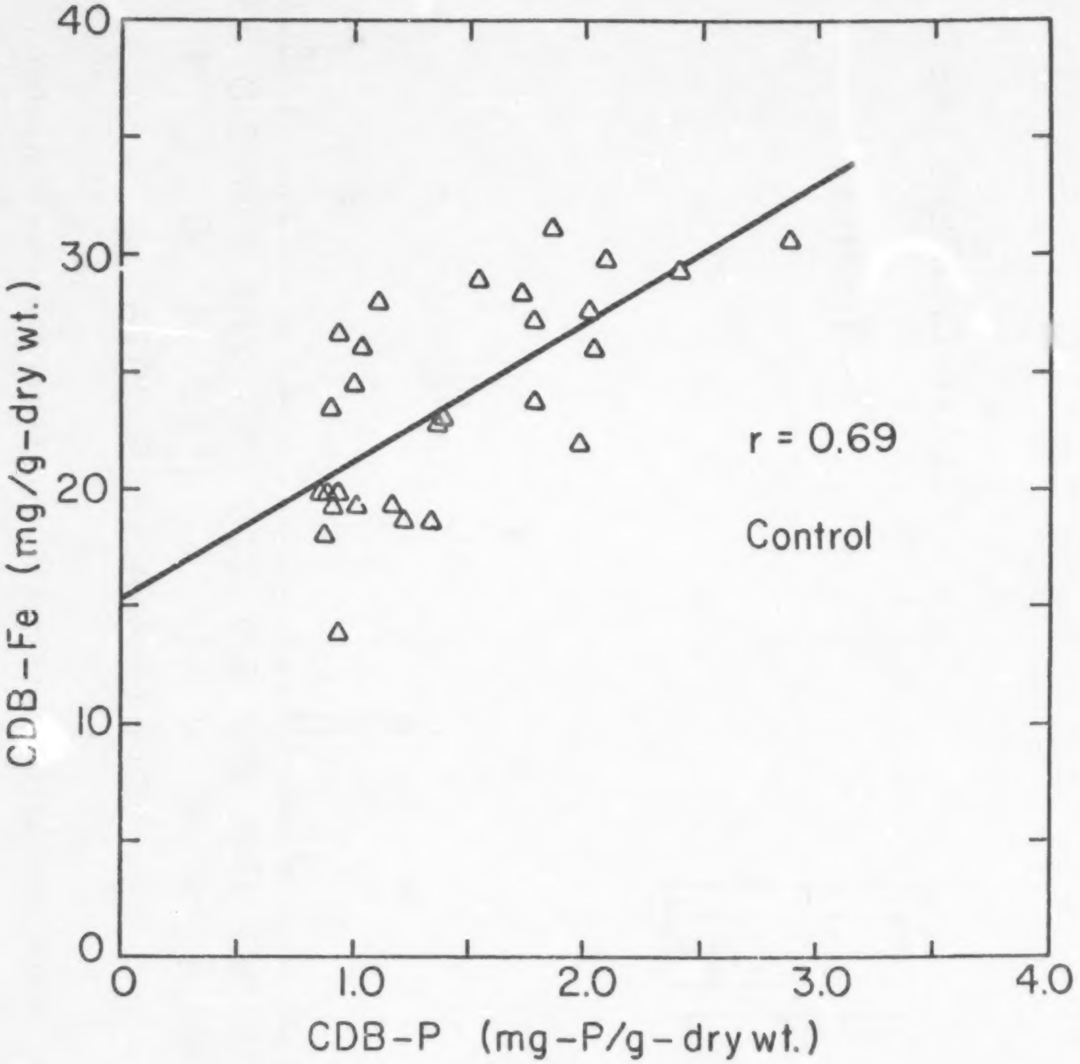

Figure 23

CDB Iron Regressed on CDB Phosphorus for Untreated Sediments.

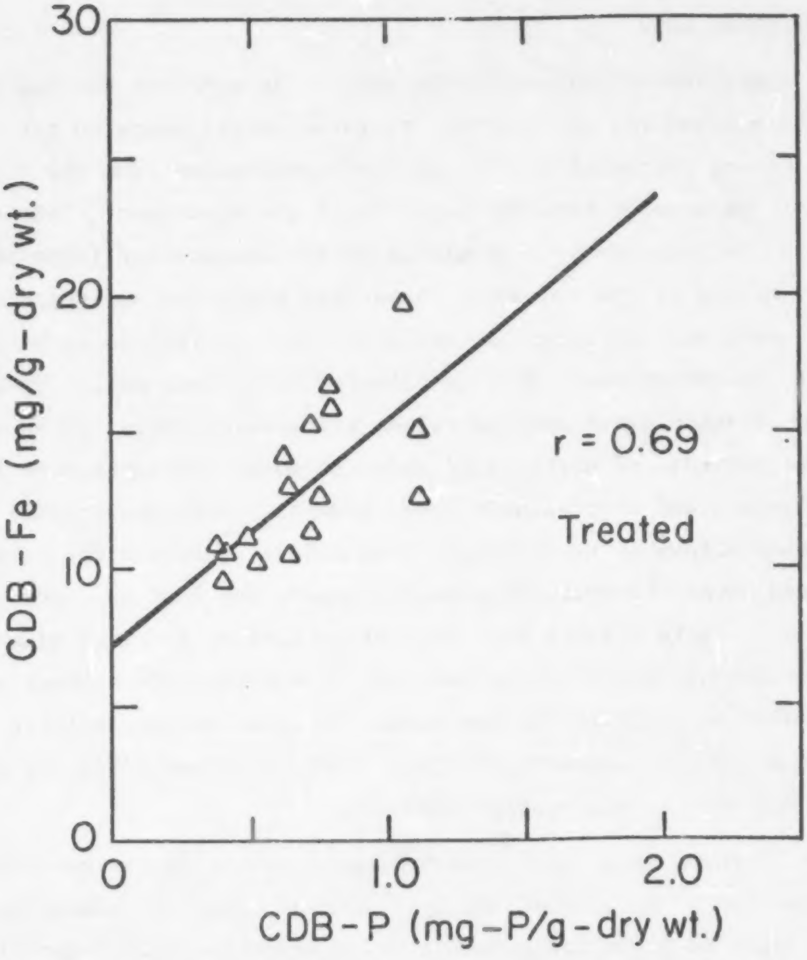

Figure 24

CDB Iron Regressed on CDB Phosphorus for Treated Sediments.

presented in Figure 25 and 26 as fractional composition plots of phosphorus forms. These diagrams are derived from averages over the depth of the cores which were analyzed. The smaller fraction of CDB-phosphorus for the treated sediments is evident. The expanded area of other inorganic phosphorus forms may be due to the lime and aluminum components of the fly ash exerting an effect on phosphorus distribution.

HEAVY METAL DATA

Legitimate concern exists in the use of fly ash over the heavy metal content of the ash and the availability of these constituents to the environment. A significant degree of cycling of toxic components from the fly ash would detract considerably from the benefits of the treatment. This concern must, of course, be balanced by a knowledge of the amounts and forms of background heavy metals in the sediments themselves since the sediments of many lakes which have been affected adversely by cultural influences have sizeable metal concentrations. This is illustrated to some extent by the data in Table 16 which gives average values for several cores, taken randomly during anoxic periods, of nitric acid digested metal concentrations in the top five centimeters and also soluble metal levels in the interstitial waters of cores. Sediment values of heavy metals show modest increases for most metals for the treated cores, however, manganese, copper, and lead concentrations exhibit a decrease. Lake Charles East has, historically, received sizeable inputs of copper for control of algae and lead from automobile exhaust and for weed control. In spite of the increases for other metals, soluble metal concentrations in the interstices of the treated sediments are not statistically distinguishable from control sediments.

Table 17 shows data for filterable heavy metals in the water column of Lake Charles East. Because of the small amount of pre-treatment data (i.e., before 5-20-76) it is not possible to draw statistically significant conclusions about the effects of fly ash on the concentrations of heavy metals in the water. The major correlation which was found was between lead concentration during the Summer and Fall 1976 and traffic density across the interstate highway bridge which bisects the lake[56]. Arsenic values seem high and are probably reflective of the previous use of lead arsenate for aquatic weed

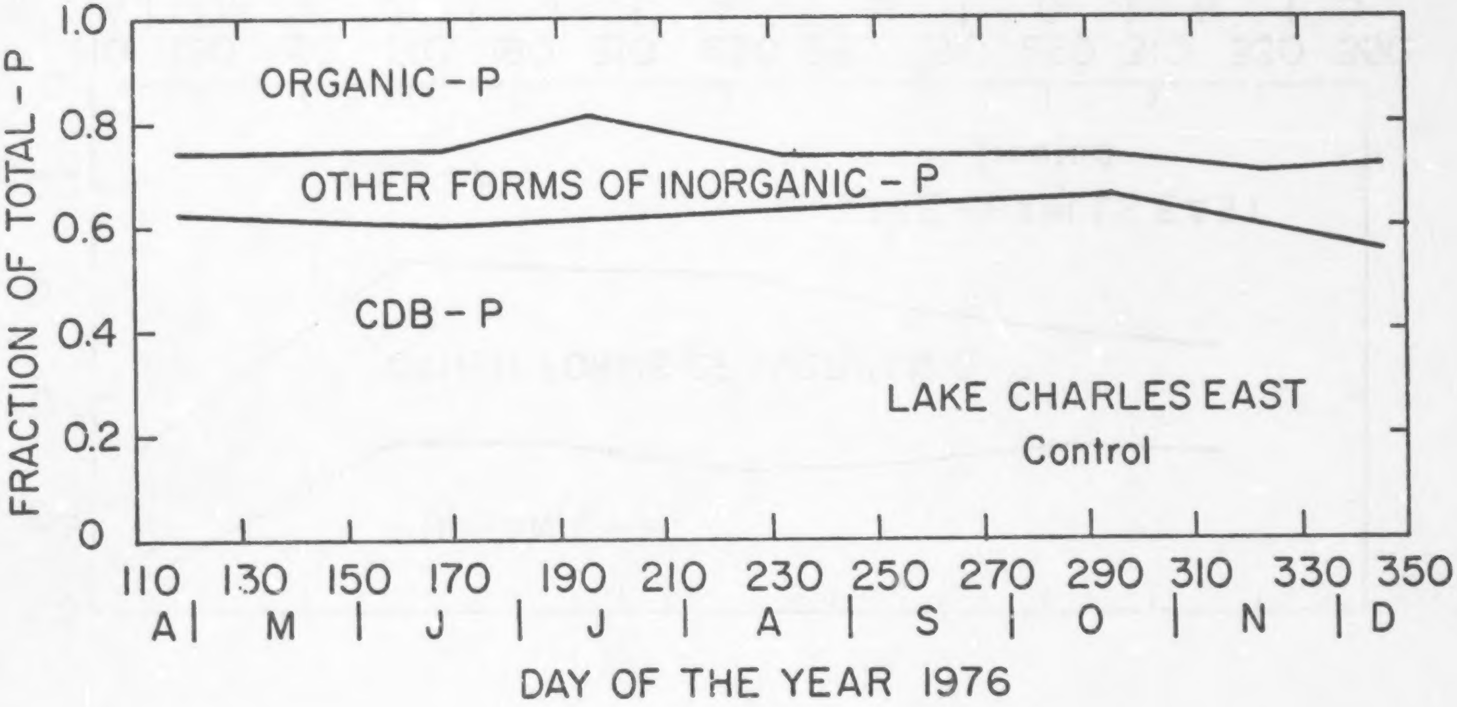

Figure 25

Fractional Composition of Phosphorus Forms in Untreated Sediments.

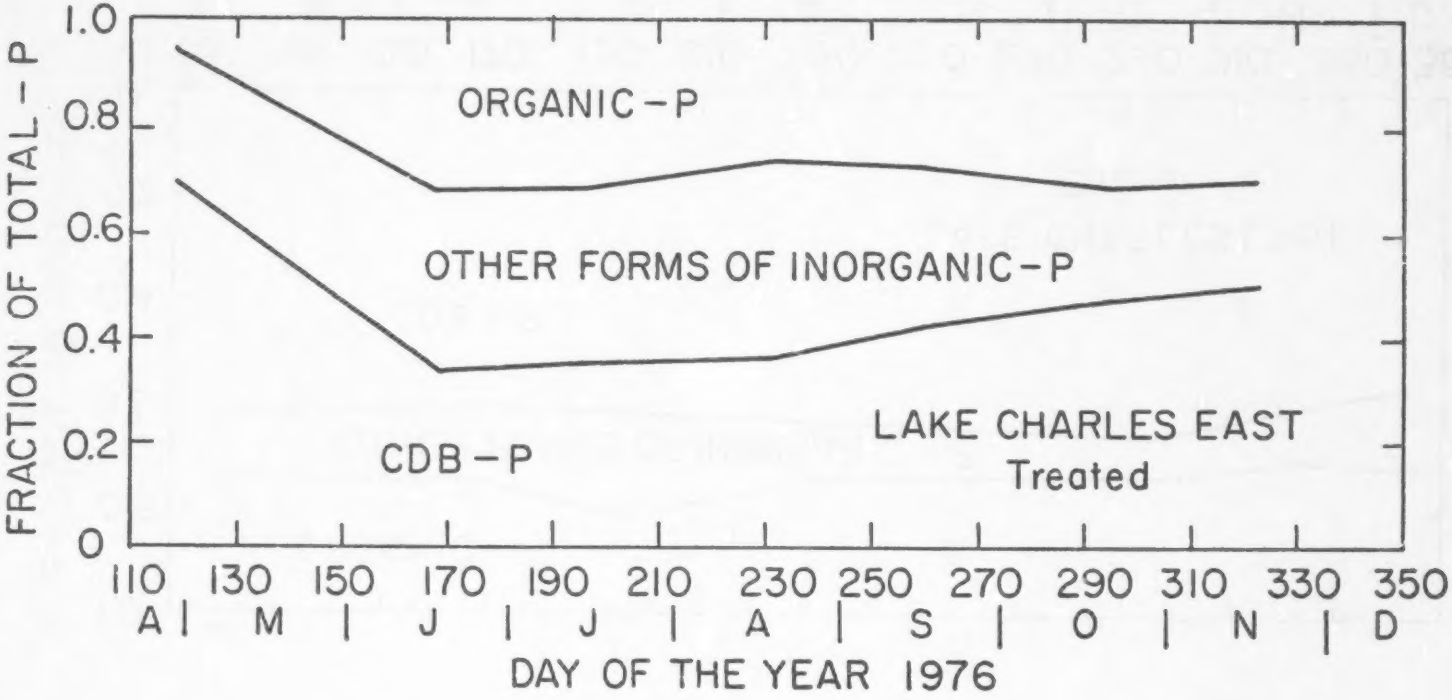

Figure 26

Fractional Composition of Phosphorus Forms in Treated Sediments.

control. Elevated concentrations of most metals in the Spring of 1976 could be due to runoff caused by spring rains. Elevated copper concentrations during periods of lake circulation are likely caused either by release from fly ash or release from sediment material. As indicated, copper sulfate was commonly used for algal control. The higher than normal nickel concentrations could be caused by aerosol deposition and/or runoff from truck traffic on the highway. Nickel is a common contaminant in diesel oil. Many of the iron concentrations are in excess of that predicted from solubility calculations for Fe (III) in an oxygenated environment. Occassional analysis for Fe (II) showed nearly half of the iron in this reduced form suggesting complexation by soluble organics plays a role in the iron chemistry of this system.

The data of Tables 16 and 17 are site and fly ash specific. Careful evaluations of increases in metal concentrations must be made for each individual type of ash material and lake sediment. Of great importance in assessing the overall problem of metal availability is the specific chemical form of the metals, whether present initially or added with the fly ash. It is suggested that a bioassay test could be developed using a sediment dwelling organism since such a creature is likely to be the most sensitive to metal uptake from sediments and is probably the most immediate pathway for the entry of metals into the aquatic food chain.

TABLE 16

Average Metal Concentrations in Sediments and Pore Water for Treated and Control Samples (n = 14)

Metal	Sediments(µg/g)		Pore Water(µg/ℓ)	
	Treated	Control	Treated	Control
Cd	3.0	1.4	2.1	3.8
Cr	29	23	3.0	3.5
Cu	54	66	21	23
Fe	7400	3700	1690	1680
Mn	1000	1490	1380	1320
Ni	152	132	78	81
Pb	140	183	82	69
Zn	-	-	13	16

TABLE 17

Trace Metals in the Water Column of Lake Charles East
(all values μg/ℓ)

	Fe	Mn	Cu	Cr	Cd	Pb	Zn	Ni	As
4-05-75(n=4)	32.5	3.0	2.9	2.2	4.3	39.0	1.8	37.0	96.0
5-20-75(n=3)	21.1	68.3	3.6	1.2	4.4	39.1	1.0	42.3	110.7
8-05-75(n=7)	43.3	101.9	3.6	1.4	4.4	56.9	4.5	113.6	84.6
4-14-76(n=8)	226.0	13.9	37.5	16.0	0.3	33.6	9.3	383.8	112.0
5-13-76(n=8)	250.0	16.9	25.5	17.6	0.2	32.0	6.1	410.0	107.5
5-26-76(n=4)	78.8	28.1	4.8	4.3	0.6	42.9	1.4	53.0	61.5
6-16-76(n=4)	254.6	33.8	4.3	12.4	0.7	56.5	1.1	87.5	66.0
6-30-76(n=4)	130.0	41.3	5.0	1.3	3.6	60.0	3.7	110.0	54.5
7-14-76(n=4)	171.3	83.1	9.0	6.0	3.1	61.0	4.4	41.8	58.0
7-28-76(n=4)	96.3	90.8	7.6	1.2	3.3	55.0	3.7	36.3	59.8
8-11-76(n=4)	130.0	42.4	7.9	<0.4	0.8	90.0	5.0	30.0	56.5
8-25-76(n=4)	215.0	118.9	5.1	4.5	0.9	118.8	7.5	83.8	68.0
9-15-76(n=4)	296.3	179.4	38.8	4.4	2.8	58.0	7.4	93.1	61.0
10-04-76(n=4)	538.8	495.0	27.5	6.8	2.9	25.5	4.9	70.5	56.0
10-20-76(n=4)	576.3	345.6	31.3	5.3	1.7	34.3	4.8	52.5	51.0
11-17-76(n=4)	537.5	92.5	9.5	3.0	10.1	111.5	5.0	70.0	43.0
12-09-76(n=4)	518.8	185.0	6.5	3.0	4.2	111.5	2.3	63.8	40.0

SECTION VIII

ANALYSIS OF DATA

BIOLOGICAL DATA

Phytoplankton Analysis

Figure 11 and 12 in Section VII illustrate the effect of diversion followed by liming and fly ash treatment on phytoplankton in Lake Charles East. Figure 11 clearly demonstrates a qualitative change in phytoplankton composition following the treatment. The lake was dominated by blue-green algae for the entire year prior to treatment in the Summer of 1975. The early part of 1975, saw a burst of chlorophyte and diatom growth, but the system was still dominated by cyanophytes. As the treatment was completed, the blue-greens experienced a decline in importance. This decline was coincident with the appearance in the lake of the cryptophyte, *Cryptomonas*, and a bloom of diatoms. *Cryptomonas* dominated the phytoplankton biomass until late May 1976, when blue-greens regained their importance and dominated through the summer. *Cryptomonas* reappeared in the end of 1976, and retained its importance in the phytoplankton until the end of sampling in 1977.

The blue-green bloom species were different in the before- and after-treatment samples. Prior to treatment the dominant algae were *Oscillatoria agardhii*, *O. putrida* and *Anabaena circinalis*. After treatment, the dominant blue-greens (during the restricted bloom), were *Anabaena circinalis*, *Microcystis aeruginosa* and *Oscillatoria agardhii*, respectively. There also occurred an increase in algal diversity following treatment of the lake. Diatoms were represented by a more sustained population through the year. Dinoflagellates (particularly *Gymnodinium*) appeared in samples for the first time in the summer following treatment. The green flagellates (specifically *Trachelomonas*) became less important in the phytoplankton relative to diatoms following the lime and fly ash application.

There was apparently also a change in biomass of phytoplankton with respect to before- and after-treatment samples (see Figure 12). As indicated previously, Lake Charles East was in a state of almost continual bloom conditions. The summer bloom of 1975 (year of the treatment), was possibly the result of nutrients suspended during disruption of the bottom sediments while the lake was being treated. In any event, as indicated previously, there was a significant change in the lake between the Summers of 1975 and 1976. While a summer bloom took place in 1976, the major species were different than in the previous year. Algal biomass in the interim period fell below the level observed during the same time in the year prior to application of fly ash. This trend was repeated in the first half of 1977. Figure 13, secchi disc transparency, shows the increasingly positive effects on the clarity in the water column from 1974 (before diversion and treatment) to 1977.

In summary, it might be said that phytoplankton in Lake Charles East were affected in both quantity and quality, with biomass generally decreasing and diversity increasing after wastewater diversion and treatment. The successional pattern which appears dominant is the more normal progression of diatoms - green algae - blue green algae. The blue greens, rather than dominating essentially year round, are confined to a late summer bloom period. The changing nutrient levels in the system during this period would appear to be primarily responsible for this alteration in phytoplankton succession.

Zooplankton Analysis

The Rotifer Community

As indicated previously, rotifers in Lake Charles East exhibit two seasonal population peaks. The species compositions at times of high densities are different for each year (see Figure 14, Section VII). The relatively diverse rotifer faunas found in the spring peaks of 1975 and 1976, are in contrast to that of 1977. The fall peak of 1975 has more evenly distributed species populations than either 1974 or 1976. In general, the spring fauna is more diverse than that of the fall. There were negligible rotifer populations in the winter of 1975-76 and 1976-77; but in the Winter of 1974-75, there was a large population of *Polyarthra dolichoptera*. The high densities of rotifers found in the warmer months of each year seems to coincide with the high algal biomass also found at that time.

Although the rotifer community seemed to be very regular and predictable in terms of time of occurrence and abundance, the same cannot be said about the individual species within the community itself. The species involved have a much wider population fluctuation in any given year, and between years. Four categories of rotifers were recognized based on their time and regularity of appearance in our samples. The first category includes species that appear regularly each year, and include *Anuraeopsis fissa*, *Brachionus angularis*, *Filinia longiseta*, *Keratella quadrata*, *Polyarthra remata* and *Proalides* sp. The second category appeared only in 1975 and 1976. It includes *Asplanchna girodi*, *Brachionus havanensis*, *B. quadridentatus* and *Pompholyx complanata*. These species achieved very high population densities in either one or both years. The third category contains species that made their first appearance in 1976, and includes *Brachionus calyciflorus*, *Euchlanis* sp., *Hexarthra mira*, *Keratella cochlearis*, *Lecane* sp., *Philodina* sp. and *Trichocera stylata*. The unidentified *Philodina*, *Lecane* and *Euchlanis* species, together with *Hexarthra mira* were present only in 1976, but *Trichocera* and *Brachionus calyciflorus* were present in both 1976 and 1977. The last category included *Lepadella patella* and *Polyarthra dolichoptera* that appeared in the Winter of 1974-75, *Filinia cornuta* in the Fall of 1974, and *Brachionus urceolaris* in the early spring of 1977.

Although the average summer algal biomass in 1975 was higher than in 1976, it did not result in higher rotifer diversity. Instead, the high rotifer diversity of 1976 is probably the result of high algal diversity seen in that year.

The Cladoceran Community

High densities of the cladoceran *Bosmina longirostris* were observed in the late Summer and Fall of 1974 and 1976, but not in the Fall of 1975 after the chemical treatment of the lake. Daphnia longiremis and D. ambigua appeared in the Fall of 1974, 1975 and 1976 with extremely high densities in the Winter and Spring of 1975-76, and exhibited wide fluctuations in the Summer of 1976. Unusual densities of one Alona species, Ceriodaphnia quadrangula and Chydorus sphaericus were also detected in the Spring and Summer of 1976. Although the exact causes for the observed fluctuations could not be determined, plausible explanations can be offered in view of available data on the algal community of the lake and literature on cladoceran populations in the other studies.

After the chemical treatment of the lake, the algal species composition changed dramatically from a blue-green algae dominated association to one dominated by flagellates. The flagellates prevailed from October 1975, to late June 1976. It was during the same period that the Daphnia species experienced the highest density. It is likely that the shift of algal composition induced the increase in Daphnia. Supporting evidence from other studies also indicated that small algae, such as many flagellates, diatoms and some green algae, are more nutritious than filamentous blue-green algae such as Anabaena and Oscillatoria[39,40,41]. In the Fall of 1974 and 1976 Daphnia species again peaked in the presence of desirable algal species in the lake. All of these results strongly suggested that the quality of food resource is limiting the growth of the Daphnia populations. The decline of Daphnia in February 1977 was not due to changes of algal food since green algae and diatoms continued to dominate the algal community at that time. Rather, the demise could be attributed to the extremely low oxygen content of the lake water during that period, for it is well-known that most cladocerans require relatively high dissolved oxygen levels in order to survive[47].

Fish predation of cladocerans in lakes had been suggested in the literature to be an important regulatory factor on crustacean community dynamics[42,43]. However fish predation has been shown to be size-selective and operates only on large cladocerans. The two Daphnia species found in Lake Charles East are quite small, and their fluctuation in the lake seems unlikely to be controlled by fish predation.

There are many indications in the literature that Bosmina species are poor competitors because of their slow developmental rate[44] and can consume only restricted sizes of food particles[45]. One behavioral adaptation for survival which was suggested by Kwik and Carter[44] was that Bosmina readily migrates to the bottom and switches from filtering of plankton to grazing of microbenthos at the water-sediment interface. Another adaptive strategy would be their remaining pelagic, but using a less preferable planktonic food resource when competition is imminent. The hypothesis that the Daphnia species in Lake Charles East are highly inefficient in utilizing blue-green algae, and therefore, leave this food resource open for Bosmina to exploit seems to be the explanation for the presence of B. longirostris in the lake. Population peaks of B. longirostris in Lake Charles East were always associated with dominance of blue-greens in the algal community. Similar relationships for these two species have also been observed by Allen[46] in Frains Lake, Michigan, where a Daphnia bloom was coupled with the abundance of phytoflagellates from February to April, while Bosmina longirostris increased in numbers only when blue-greens and a large dinoflagellate became dominant in late May.

The rest of the cladoceran species, which included one species of Alona, Ceriodaphnia quadrangula and Chydorus sphaericus, were present in the lake sporadically in low numbers in 1974 and 1975. In the Spring and Summer of 1976 comparatively large fluctuations in numbers were observed, and coincided with the rapid shifting of algal species from June to August of that year. In the literature it is reported that Chydorus sphaericus would abandon its benthic habitat and become planktonic in times of cyanophycean blooms[47] and summer blooms of Ceriodaphnia quadrangula and Chydorus sphaericus were also observed in a Masurian lake by Gliwicz[48]. It seems likely that the populations of these two species in Lake Charles East are also responding to food species as a controlling factor.

The long-term effect of the chemical treatment on the cladoceran community appears to be an indirect one. After the treatment, the species of primary producers was altered considerably. The increase in algal diversity in 1976 may well explain the presence of cladoceran species such as Chydorus sphaericus

sphaericus and Ceriodaphnia quadrangula. The pattern of 1976 was not repeated for the spring-summer period of 1977; therefore, it is not possible to say that the treatment had a lasting effect on the zooplankton dynamics in Lake Charles East.

The Copepod Community

Spring and fall peaks separated by summer minima characterized the copepod population encountered. Summer aestivation of many cyclopoid species in the copepodite IV and V stages has been reported[49,50,51,52] and might account for the disappearance of the nauplii in the summer months at Lake Charles East. The breaking of summer diapause and resumption of normal development were reflected in the high fall adult copepod populations of 1974 and 1976.

Development of the copepod community seemed to be normal for the early part of 1975, when a nauplius peak was observed. However, the summer minimum of 1975 coincided with the time of chemical treatment of the lake, and was not followed by the reappearance of either peaks of nauplii or adults in late summer. The increase of nauplii was delayed until the winter months of 1975-76. Though there was no direct verification, it is possible that this peculiar behavior of the copepod community in 1975 was caused by the chemical treatment in the summer of that year. The amount of fly ash and lime mixture applied to the lake put a considerable layer on the lake bottom. This layer undoubtedly buried the aestivating copepodites and might later have constituted a physical barrier which prevented the copepodites from re-entering the lake after breaking diapause. The chemical environment surrounding the aestivating copepodites could also be altered by the fly ash and lime mixture and become detrimental to them. Repopulation of the lake by copepods after the chemical treatment was much slower than that of the rotifers or cladocerans. Since the rest of the planktonic community rebounded rapidly from the treatment, the copepods were probably not handicapped by lack of food.

Any long-term effect of the chemical treatment on the copepod community is not apparent in this study. The annual cycle, disrupted in 1975, was resumed in 1976 and 1977, and population densities of 1976 and 1977 seemed comparable to those of 1974 and 1975 prior to treatment.

Fish

Other than the immediate fish-kill described in Section VII of this report, no long-term effect could be noted with respect to the fish population. Evidently the great majority of fish, especially the bass, were able to avoid the area of the lake being treated at any particular time. In this way, they were able to minimize the acute effects of the lime and fly ash. The fish kill was probably due to the rapid rate at which the final lime application was carried out. There simply was no safe area of the lake in which they could take refuge. This suggests that fish kills might be averted so long as the area under treatment at any given time was strictly controlled. As mentioned earlier, only small bluegills were observed as being affected by the kill.

SEDIMENTS

The demonstrated effectiveness of the fly ash in reducing the release of phosphorus during anoxic periods (Figures 19 and 20) would appear to be due to two factors: formation of a physical barrier and changes in the chemical properties of the ash-sediment material. To the extent that phosphorus release is a diffusion controlled reaction in lake sediments[7,12,6] it could be expected that the finer grain size contributes to a decreased permeability thereby increasing the diffusional path length or tortuosity for the ash layer. The average moisture content of the fly ash layer was measured to be 1.79 mg of water/gram dry weight of sediments as compared to 18.77 mg/g for the top centimeter of the control sediments. However, the generally spherical nature of ash particles would work counter to this trend in comparison to the control sediments which possess a large clay fraction and therefore a much lower degree of "sphericity". In situ measurements of permeability for the two sediment types are difficult and were not attempted in this research.

In any case, it appears that the chemical alterations brought about by the fly ash are also of considerable importance in assessing the effects noted. The specific fly ash used in this study had a relatively low lime content (Table 11) and the data presented suggest strongly that the resulting lower amorphous iron content (Figures 23 and 24) is largely responsible for the lower amounts of phosphorus in a form available for release (Figures 21 (b) and (c); 22 (b) and (c)). Both CDB iron and phosphorus values, measures of the capacity for and amount of phosphorus available for release, were reduced by an average 37% and 44%, respectively, in the treated sediments. This aspect of the results should be noted in addition to the observations of Yaksich[25] and Tenney[18] in which the lime content of the ash was postulated as the important phase controlling the chemistry of phosphorus.

The lower CDB-iron content of the treated sediments raises questions concerning the effectiveness of different fly ashes in sealing eutrophic lake sediments. Theis and Wirth[53] have shown that the relative amounts of amorphous iron oxides and lime are a good indication of the ultimate acid-base character of a fly ash. It seems unlikely that an acidic ash could be of use in the treatment of lake sediments, however, many basic fly ashes contain large amounts of CDB-iron, the normally acidic character of this component being masked by large amounts of lime. If CDB-iron is viewed as the primary phosphorus sink material, then fly ashes which have low amounts of this phase should be favored for use in lake sediments.

The development of the detrital layer above the fly ash suggests the need for some measure of control over lake production of primary producers. Residual phosphorus levels after treatment in Lake Charles were sufficient to promote excessive algal activity. The resulting re-establishment of essentially pretreatment type sediments above the ash was very likely the major source of phosphorus released by the treated sediments during anoxic periods. Since an annual treatment of lake sediments with fly ash is probably not feasible, there would appear to be two approaches to the problem. Addition of a supplemental chemical, such as lime or alum, to the lake in sufficient quantities and with the necessary degree of mixing and dispersion could bring about the desired lower levels of phosphorus to limit growth. Cooke and Kennedy[54] have used alum effectively for this purpose. Alternately, a suitable algacide, such as cupric ion, could be used until ambient phosphorus levels have been sufficiently reduced through natural lake flushing.

CHEMICAL MODELING

Figure 7 indicates a loss of total phosphorus from the water column in Lake Charles East after diversion of wastewater and treatment of sediments. Analysis of sediments has shown that the fly ash was effective in retarding the release of phosphorus during anoxic periods. The loss of phosphorus from the lake is due to two factors: flushing and interchange with the sediments. For purposes of this study, it is important to assess the relative contribution of the fly ash treatment to the phosphorus budget of the lake. This is most conveniently approached through an approach which makes use of mathematical models which describe the system.

There are basically two types of lake models, those which include seasonal phytoplankton interactions and distinguish between total and soluble forms of phosphorus, and phosphorus budget or input-output models which generally predict total phosphorus only. It is this latter type of model which is applied to the data of Lake Charles East.

The phosphorus vs. time data for Lake Charles East (Figure 7) can be fitted rather well by an exponential decay function. This suggests that a simple input-output model in which the lake is treated as a completely mixed reactor with a phosphorus reaction term should be applicable. The phosphorus residence approach of Sonzogni and Uttormark[55] is such a model and will be used initially here. Basically, the model consists of a mass balance on phosphorus.

$$V \frac{dC_p}{dt} = QC_p^i - QC_p - kC_pV \qquad \text{VIII-1}$$

where

V = lake volume (assumed constant),

Q = flow,

C_p = concentration of phosphorus,

C_p^i = influent phosphorus concentration, and

k = internal phosphorus reaction rate constant.

The internal loss constant, k, is assumed first order with respect to the phosphorus concentration. It thus has units of time^{-1}.

If the lake is well mixed, equation 1 can be integrated to

$$C_p = C_p^{\infty} - (C_p^{\infty} - C_p^{o})e^{-t/R_p} \qquad \text{VIII-2}$$

where

C_p^{∞} is the steady state phosphorus concentration,
C_p^{o} is the initial phosphorus concentration at time zero, and
R_p is the phosphorus residence time which is equal to $(1/R_w + k)^{-1}$,
R_w = hydraulic residence time, V/Q.

In the original model formulation, k was considered to be a positive quantity reflecting loss of phosphorus to the sediments (through the mechanisms of sedimentation and adsorption). It was taken to be constant reflecting, at least over the short term, the specific chemistry of phosphorus in a lake system. Under these conditions, the phosphorus residence time is always less than or, when k is zero (i.e., phosphorus behaves conservatively) equal to R_w. It should be noted, however, that there is no intrinsic reason why k may not vary or assume a negative value (the absolute value of which is not greater than $1/R_w$) for periods of excessive phosphorus release from sediments, or cycling of phosphorus in the water column. In such cases, R_p is greater than R_w.

If a sufficiently long time base is used (i.e., months to years), the calculation of k and hence, the phosphorus residence time is straightforward. Using equation 1 and approximating the differential linearly, one obtains

$$k \; \frac{V\frac{\Delta p}{\Delta t} - P_{in} + P_{out}}{V\,P_{avg.}} \qquad \text{VIII-3}$$

where phosphorus quantities are on a total mass basis. Equation 3 can be applied to any comprehensive set of phosphorus budget data.

This is done for the data of Lake Charles East on a yearly basis from 1973 through 1977 in Table 18. In all years except one, 1975, the loss constant is positive indicating loss to the sediments was greater than release from them. 1975 was the year after external abatement of phosphorus loading and the year of sediment treatment. The loss constant reflects a definite change after abatement suggesting the sediments play a much more important role when external phosphorus loading is comparatively small.

The variation in sign and magnitude of k becomes more pronounced when calculations are made on a seasonal basi in Lake Charles East. Data are given in Table 19. Here the time period 1973 to 1977 is divided decimally so that there are approximately two calculation periods per year corresponding roughly to the winter and summer seasons. Several important observations can be made from these data. It is clear that before abatement (which occurred at 1974.8) the net flux of phosphorus is strongly toward the sediments. After abatement and treatment the net is still toward the sediments (Table 18) but the annual cycle of uptake and release is more balanced. Thus, the comparatively small internal reaction rate constants for 1976 and 1977 calculated from the yearly time interval are seen to consist in reality of a relatively large set of oppositely paired fluxes, one toward, the other from the sediments. The relative magnitudes of each one of the pair are nearly the same giving rise to a reaction rate constant on a yearly basis which is very small. This cyclic variation is in concert with the observed uptake and release trends in the sediments and offers an explanation for the oscillation of phosphorus levels observed in the water column, visible in Figure 7. The initial conclusion which could be inferred from Table 18 is that phosphorus behaves as a chemically conservative substance ($R_p \cong R_w$) whereas the data of Table 19 more accurately reflect the known reactivity of phosphorus in lake systems.

The fact that peak phosphorus concentrations in the water column within a yearly cycle and maximum release from sediments do not coincide suggests that algal uptake and sedimentation play an important role in Lake Charles East. The algae take up large quantities of phosphorus from the lake and subsequently settle to the sediments. Fall circulation brings about a large increase in total phosphorus. Chemical precipitation or adsorption onto

TABLE 18

Calculation of Internal P Loss Constant - Yearly Basis
Lake Charles - 1973-1977

					A		B	C	D			
Year	P concentration (mg-P/ℓ) Initial	Final	Ave.	Δ P g/m^3	Δ P stored g/yr.	Q_{out} $m^3/yr.$	P_{out} g/yr.	P_{in} g/yr.	V $P_{avg.}$ g	$k = -(\frac{A-C+B}{D})$	R_w years	R_p years
1973	1.38	0.93	1.15	-0.47	-92,120	117,000	134,600	208,714	225,400	+.74	1.7	.75
1974	0.93	0.60	0.76	-0.31	-60,760	78,800	59,900	143,714	148,000	+.97	2.9	.76
1975	0.60	0.40	0.50	-0.20	-39,200	132,000	66,000	7,714	98,000	-.21	1.5	2.19
1976	0.40	0.27	0.34	-0.13	-25,480	77,600	26,400	12,110	66,600	+.06	2.5	2.17
1977	0.27	0.18	0.23	-0.09	-17,640	83,600*	19,200*	4,710*	44,100	+.07	2.3	1.98

*Estimate

TABLE 19

Calculation of Internal P Loss Constant - Seasonal Basis
Lake Charles - 1973-1977

				A	B	C	D	
Time Period	Tenths of Years	Δ P mg-P/ℓ	$P_{avg.}$ mg-P/ℓ	V $P_{avg.}$ g	Δ P stored g	P_{in} g	P_{out} g	$k=-(\frac{B-C+D}{A})$ (yr^{-1})
1973.8-1974.5	7	-0.83	0.85	166,600	-162,600	122,700	59,570	+1.35
1974.5-1974.8	3	+0.24	0.69	136,000	+ 47,000	46,890	18,900	-0.14
1974.8-1975.4	6	-0.66	0.58	114,000	-219,360	39,370	29,460	+1.22
1975.4-1975.8	4	+0.35	0.47	92,600	+ 68,600	8,110	26,800	-0.94
1975.8-1976.5	7	-0.38	0.38	74,500	- 74,500	16,370	25,500	+0.88
1976.5-1976.7	2	+0.12	0.32	61,740	+ 23,500	4,940	4,820	-0.38
1976.7-1977.4	7	-0.16	0.26	50,960	- 31,400	14,690	12,950	+0.65
1977.4-1977.6	2	+0.16	0.22	43,120	+ 31,400	3,460	3,340	-0.73

sediments or settling oxide or clay particles could also be contributory mechanisms.

It has been shown that the release of phosphorus from Lake Charles East sediments during anoxic periods is significantly reduced by the fly ash layer. Thus, the negative values of k in Table 19 after treatment are inclusive of this retardation, albeit the effect may be small since only 35% of the sediments were treated and the constant is calculated from the data for the entire lake basin. Nevertheless, based upon sediment release values measured, it is possible to model the long-term effects of the abatement/treatment restoration scheme using the basic input-output model described. Results of this effort are shown in Figure 27 in which total phosphorus is plotted as a function of time. Since both R_w and k were allowed to vary, a simple closed form solution to the model could not be used. Rather, time varying solutions to the mass balance equation were obtained using numerical integration techniques. Figure 27 contains computer simulations of five possible scenarious for Lake Charles East:

1) Unabated phosphorus inputs to the system at essentially pre-1972 levels.
2) Unabated phosphorus inputs in which Indiana's detergent phosphate ban is included. Phosphorus loading at 1972-1974 levels.
3) Abatement of external phosphorus sources to the degree experienced in mid-1974 (approximately 90% reduction).
4) Same as 3) with treatment of 35% of the lake sediments with fly ash. This combination of manipulation techniques is the course actually followed in Lake Charles East.
5) Same as 3) with treatment of all sediments which release phosphorus.

Actual data points are also shown in Figure 27. The degree of fit between scenario 4 and the historical data lend credence to the future projections which are given.

Figure 27 predicts a steady state total phosphorus concentration in the lake of 0.23 mg-P/ℓ. This is 0.06 mg/ℓ less than if no sediment treatment had taken place and still considerably above the level considered necessary to limit phytoplankton growth to non-bloom conditions. It thus appears that for the foreseeable future Lake Charles East will remain a eutrophic system.

Nevertheless, there are documented improvements in overall water quality consisting of altered phytoplankton composition in which noxious blue-green algae are more limited, reduced total biomass, and increased water clarity for large portions of the year. In assessing the relative improvement of a culturally hypereutrophic system such as Lake Charles East it is apparent that the standard ranges of health indices normally applied to less polluted systems do not portray accurately the extent of change. For example, it appears that blue-green algae are not able to compete as effectively with flagellated species at 100 μg-P/ℓ. Aspects of lake pollution such as this are not as well documented as they might be and reflect, perhaps a lesser amount of research activity on these types of lake systems.

With respect to the covering of lake sediments to reduce internal phosphorus loading, it should be noted that Figure 27 predicts a steady state phosphorus concentration of 0.11 mg-P/ℓ in Lake Charles East if 100% of the sediments are treated. This is a reduction of 61% over the level predicted if no sediment treatment had taken place.

The suitability of sediment covering with fly ash or other particulate materials to improve water quality for a given lake depends upon several factors. These would include the physical and chemical properties of both sediments and material especially as they impact on phosphorus chemistry, the relative contribution of sediments to the phosphorus budget of the system, lake morphology, the need for supplemental chemical addition, possible deleterious effects such as fish kills and heavy metal cycling, and aesthetic and economic factors. Although it was not within the scope of this research to perform a detailed economic analysis of the sediment treatment, it is obvious that transportation costs of the particulate material would very likely be a major factor. Bulk hauling rates range from $.05 to $.20 per ton-mile depending upon the type of vehicle used.

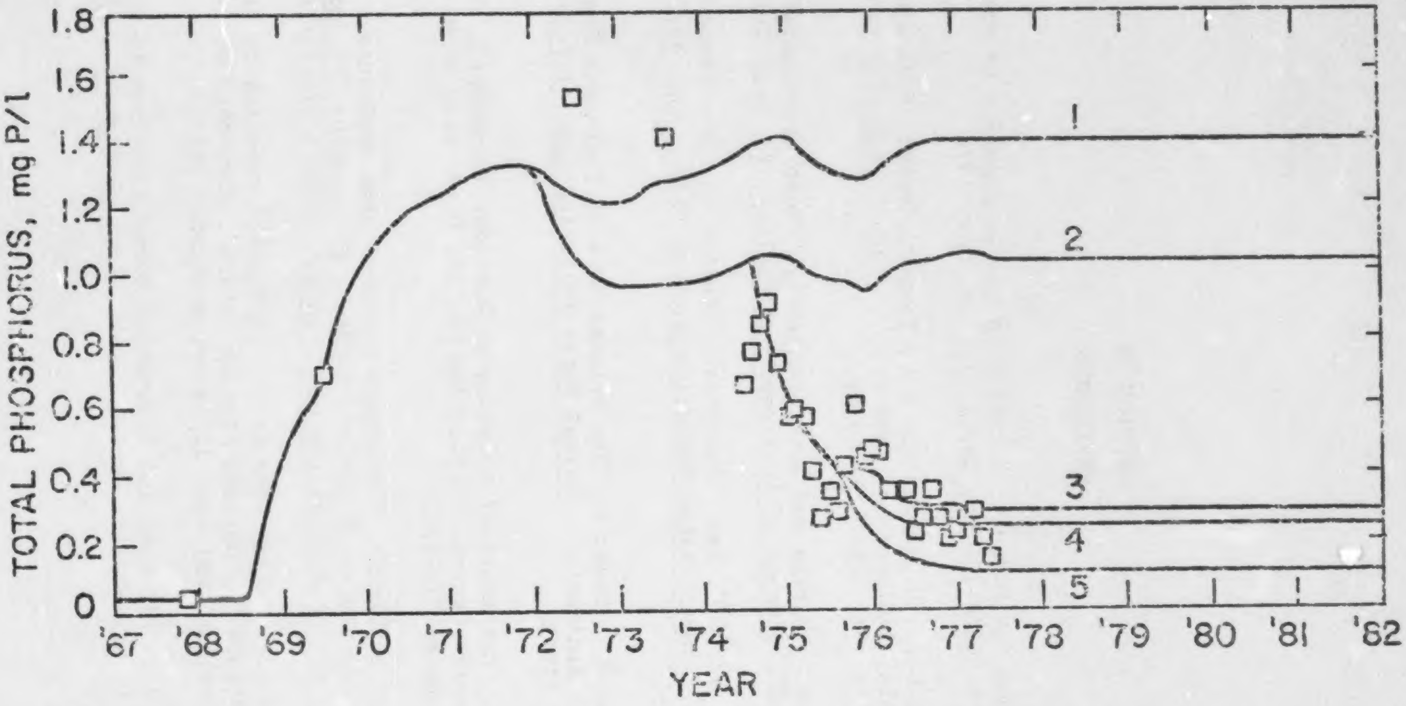

FIGURE 27: Input-Output Modeling of Phosphorus in Lake Charles East (numbers refer to scenarios given in the text).

SECTION IX

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APPENDIX

ZOOPLANKTON COUNTS IN LAKE CHARLES EAST

Following are zooplankton populations in Lake Charles East for use with the discussions presented in Chapters VII and VIII. Dates are given in consecutive numerical order beginning with January 1, 1974. Thus the first day of monitoring was number 189 (189th day of 1974). The abbreviations given below are used for each genus or species of organism found.

Rotifers

Anura	Anuraeopsis fissa
Br3	Brachionus calyciflorus
Br4	B. quadridentata
Spla	Asplanchna
Euchl	Euchlanis
Br5	Brachionus urceolaris
Br6	B. havanaensis
Hex	Hexarthra
Phil	Philodina
Bangul	Brachionus angularis
Poly	Polyarthra remata
Kerqua	Keratella quadrata
Pomph	Pompholyx sulcata
Filin	Filinia cornuta
Proal	Proalides sp.
Lepad	Lepadella sp.
Trich	Trichocera sp.
Lecan	Lecane
Kcod	Keratella cochlearis
Br2	Brachionus diversiconis

Copepods

Cope	Copepod nauplii
Cyclop	Cyclops and other related genera

Cladocera

Bosm	Bosmina longirostris
Daph	Daphnia spp.
Cerio	Ceriodaphnia sp.
Chyd	Chydorus
Alona	Alona

LCE ZOOPLANKTON BY DATE

number/liter

	ROTIFERS									COPEPODS	
Date	Anura	Br3	Br4	Spla	Euchl	Br5	Br6	Hex	Phil	Cope	Cycl
189.00	1191.00	0	-0	-0	-0	-0	-0	-0	-0	397.00	198.00
203.00	979.00	0	-0	-0	-0	-0	-0	-0	-0	537.00	312.00
210.00	639.00	0	-0	-0	-0	-0	-0	-0	-0	341.00	440.00
218.00	99.00	0	-0	-0	-0	-0	-0	-0	-0	156.00	795.00
231.00	85.00	0	-0	-0	-0	-0	-0	-0	-0	238.00	369.00
257.00	9.00	0	-0	-0	-0	-0	-0	-0	-0	0	0
272.00	57.00	0	-0	-0	-0	-0	-0	-0	-0	68.00	22.00
286.00	38.00	0	-0	-0	-0	-0	-0	-0	-0	85.00	57.00
312.00	0	0	-0	-0	-0	-0	-0	-0	-0	247.00	275.00
340.00	0	0	-0	-0	-0	-0	-0	-0	-0	141.00	85.00
372.00	0	0	-0	-0	-0	-0	-0	-0	-0	132.00	370.00
404.00	0	0	-0	-0	-0	-0	-0	-0	-0	114.00	76.00
446.00	0	0	-0	-0	-0	-0	-0	-0	-0	351.00	47.00
460.00	5.00	0	-0	-0	-0	-0	-0	-0	-0	339.00	32.00
500.00	32.00	0	-0	-0	-0	-0	-0	-0	-0	357.00	49.00
522.00	1326.00	0	-0	-0	-0	-0	-0	-0	-0	16.00	30.00
533.00	5318.00	0	-0	-0	-0	-0	-0	-0	-0	56.00	43.00
540.00	4082.00	0	-0	-0	-0	-0	-0	-0	-0	69.00	109.00
547.00	3457.00	0	-0	-0	-0	-0	-0	-0	-0	23.00	153.00
554.00	14137.00	0	-0	-0	-0	-0	-0	-0	-0	15.00	69.00
561.00	14359.00	0	-0	-0	-0	-0	-0	-0	-0	23.00	15.00
569.00	11806.00	0	-0	-0	-0	-0	-0	-0	-0	122.00	7.00
578.00	5512.00	0	-0	-0	-0	-0	-0	-0	-0	46.00	69.00
593.00	1712.00	0	-0	-0	-0	-0	-0	-0	-0	2.00	30.00
539.00	331.00	0	-0	-0	-0	-0	-0	-0	-0	1.00	1.00
596.00	943.00	0	-0	-0	-0	-0	-0	-0	-0	0	0
603.00	4914.00	0	-0	-0	-0	-0	-0	-0	-0	7.00	0
617.00	0	0	0	0	0	0	-0	-0	-0	0	3.00
620.00	355.00	0	277.00	0	0	0	-0	-0	-0	0	0
636.00	1485.00	0	91.00	0	0	0	-0	-0	-0	0	0
662.00	92.00	0	0	0	0	0	-0	-0	-0	37.00	0
703.00	0	0	0	0	0	0	-0	-0	-0	120.00	0
740.00	5.00	0	0	0	0	0	-0	-0	-0	238.00	0
777.00	0	0	0	0	0	0	-0	-0	-0	258.00	0
802.00	0	0	0	0	0	0	-0	-0	-0	232.00	0
823.00	0	0	0	0	0	0	-0	-0	-0	337.00	0
836.00	0	2.00	0	0	0	0	-0	-0	-0	129.00	0
863.00	0	0	11.00	0	0	0	-0	-0	-0	11.00	167.00
870.00	0	0	0	0	0	0	-0	-0	-0	265.00	0
877.00	4.00	0	0	0	0	0	-0	-0	-0	214.00	1.00
884.00	0	0	0	0	0	0	-0	-0	-0	87.00	0
995.00	302.00	2.00	4.00	0	0	0	-0	-0	-0	22.00	0
933.00	76.00	0	0	0	0	0	-0	-0	-0	61.00	47.00
954.00	10.00	0	0	0	0	0	-0	-0	-0	214.00	48.00
940.00	5.00	0	0	0	0	0	-0	-0	-0	128.00	41.00
968.00	12.00	0	0	0	0	0	-0	-0	-0	179.00	42.00
889.00	123.00	4.00	0	0	0	0	0	0	0	117.00	102.00
897.00	260.00	4.00	2.00	0	0	0	0	0	0	40.00	38.00
919.00	2083.00	25.00	0	165.00	0	0	0	63.00	0	35.00	8.00

ROTIFERS (cont'd) | COPEPODS (cont'd)

Date	Anura	Br3	Br4	Spla	Euchl	Br5	Br6	Hex	Phil	Cope	Cycl
925.00	219.00	0	0	48.00	0	0	0	15.00	0	75.00	23.00
946.00	14.00	0	0	0	0	0	0	0	0	94.00	131.00
974.00	3.00	0	0	0	0	0	0	0	0	41.00	114.00
988.00	38.00	0	0	2.00	0	0	0	0	0	575.00	434.00
1007.00	54.00	0	0	0	0	0	0	0	0	2063.00	797.00
1023.00	20.00	0	0	0	0	0	0	0	0	1548.00	837.00
1202.00	0	0	0	0	0	0	0	0	0	91.00	211.00
1173.00	0	0	0	0	0	0	0	0	0	10.00	7.00
1142.00	0	0	0	0	0	0	0	0	0	1.00	8.00
1107.00	0	0	0	0	0	0	0	0	0	202.00	86.00
1073.00	0	1.00	0	0	0	0	0	0	0	364.00	137.00
1051.00	0	1.00	0	0	0	0	0	0	0	705.00	71.00
1235.00	0	0	0	0	0	0	0	0	0	572.00	90.00
1258.00	1813.00	0	0	0	0	0	0	0	0	102.00	2.00
911.00	1073.00	-0	1039.00	0	22.00	0	0	0	165.00	8.00	25.00
960.00	14.00	-0	-0	-0	-0	-0	-0	-0	-0	95.00	441.00
1265.00	2589.00	0	4.00	0	0	0	0	0	0	85.00	8.00
1279.00	7386.00	5.00	0	5.00	0	5.00	0	0	0	16.00	0
1295.00	0	0	0	0	0	0	0	0	0	21.00	4.00
1312.00	0	0	0	0	0	0	0	0	0	42.00	4.00
1325.00	0	0	0	0	0	0	0	0	0	38.00	4.00
1335.00	0	0	0	0	0	0	0	0	0	116.00	8.00

Date	Bangul	Poly	Kerqua	Pomph	Filin	Proal	Lepad	Trich	Lecan	Kcod	Br2
189.00	0	0	0	0	0	0	0	0	0	0	0
203.00	0	0	28.00	0	0	0	0	0	0	0	0
210.00	42.00	0	28.00	0	0	0	0	42.00	0	0	0
218.00	0	0	56.00	0	0	0	0	0	0	0	0
231.00	14.00	0	56.00	0	0	0	0	0	0	0	0
257.00	0	0	9.00	0	0	0	0	0	0	0	0
272.00	68.00	0	4986.00	0	0	0	0	0	0	0	0
286.00	38.00	0	551.00	0	0	0	0	0	0	0	0
312.00	104.00	57.00	0	0	0	0	0	0	0	0	0
340.00	0	151.00	47.00	0	0	0	1343.00	0	0	0	0
372.00	0	1738.00	18.00	0	0	0	0	0	0	0	0
404.00	0	2688.00	0	0	0	0	0	0	0	0	0
446.00	0	1491.00	0	0	9.00	0	0	0	0	0	0
460.00	0	1013.00	0	1.00	40.00	0	0	0	0	0	0
500.00	16.00	18.00	0	1.00	0	0	4.00	0	0	0	0
522.00	0	0	0	122.00	0	0	3.00	0	0	0	0
533.00	99.00	0	84.00	1530.00	0	2.00	0	0	0	0	0
540.00	1615.00	0	207.00	8268.00	109.00	0	0	0	0	0	0
547.00	2499.00	0	733.00	2476.00	58.00	5.00	2.00	0	0	0	0
554.00	15.00	7.00	1157.00	4761.00	314.00	0	7.00	0	0	0	0

ROTIFERS (cont'd)

Date	Bangul	Poly	Kerqua	Pomph	Filin	Proal	Lepad	Trich	Lecan	Kcod	Br2
561.00	53.00	38.00	3051.00	1426.00	2215.00	7.00	0	0	0	0	0
568.00	69.00	245.00	5803.00	1564.00	3158.00	0	0	0	0	0	0
578.00	184.00	0	6509.00	352.00	383.00	23.00	7.00	0	0	0	0
583.00	117.00	15.00	291.00	56.00	360.00	46.00	0	5.00	2.00	0	0
599.00	34.00	0	5.00	7.00	251.00	0	0	0	0	0	0
596.00	130.00	0	69.00	92.00	360.00	575.00	0	0	0	0	0
603.00	513.00	130.00	268.00	299.00	828.00	2422.00	0	0	0	0	0
617.00	0	0	0	0	0	0	0	0	22.00	0	0
620.00	16.00	325.00	0	0	0	687.00	1.00	0	1.00	437.00	374.00
636.00	22.00	1731.00	0	70.00	306.00	0	0	20.00	0	68.00	2558.00
662.00	0	154.00	0	0	24.00	0	0	0	0	6.00	3.00
703.00	0	27.00	0	0	0	0	0	0	3.00	4.00	0
740.00	0	5.00	1.00	0	6.00	0	0	0	5.00	2.00	0
777.00	0	1.00	0	0	1.00	0	0	0	2.00	2.00	0
802.00	0	1.00	0	0	0	0	0	0	1.00	0	0
823.00	0	6.00	0	0	0	0	0	0	1.00	0	14.00
836.00	0	29.00	0	1.00	0	0	1.00	0	5.00	1.00	244.00
863.00	0	294.00	4.00	0	20.00	0	0	0	0	98.00	2.00
870.00	0	303.00	0	0	0	0	2.00	0	1008.00	21.00	0
877.00	0	64.00	0	0	3.00	0	0	0	2585.00	66.00	1.00
884.00	0	0	0	0	0	0	0	0	399.00	0	0
905.00	206.00	614.00	64.00	1008.00	6668.00	0	4.00	254.00	58.00	272.00	18.00
933.00	0	8.00	59.00	0	5.00	0	0	16.00	88.00	30.00	0
954.00	0	3.00	0	0	12.00	0	0	1.00	86.00	2.00	0
940.00	0	3.00	0	0	9.00	0	0	0	72.00	6.00	0
968.00	0	0	0	0	0	19.00	0	0	486.00	7.00	0
889.00	7065.00	183.00	4694.00	67.00	52.00	0	0	2.00	0	8.00	0
897.00	7981.00	3094.00	1106.00	1448.00	3165.00	0	2.00	2.00	4.00	4.00	44.00
919.00	206.00	2173.00	340.00	110.00	2419.00	0	0	379.00	2.00	1813.00	910.00
925.00	708.00	1006.00	13.00	23.00	4856.00	0	0	463.00	0	1865.00	808.00
946.00	8.00	1.00	2.00	0	1.00	0	0	0	0	0	0
974.00	670.00	29.00	6.00	0	2.00	2901.00	0	1.00	0	0	0
988.00	4710.00	2.00	0	0	0	1021.00	0	65.00	0	2.00	0
1007.00	1074.00	5.00	1.00	262.00	1.00	0	0	0	0	0	0
1023.00	65.00	0	0	480.00	0	0	0	0	1.00	0	0
1232.00	3.00	0	3.00	2.00	0	0	0	0	1.00	0	0
1173.00	0	1.00	1.00	0	3.00	0	0	1.00	0	0	0
1142.00	1.00	9.00	3.00	0	1.00	0	0	0	0	0	0
1107.00	0	1.00	10.00	1.00	4.00	0	0	0	0	0	0
1073.00	0	0	8.00	40.00	1.00	0	0	0	0	0	0
1051.00	0	0	8.00	105.00	0	1.00	0	1.00	0	0	0
1235.00	17.00	0	7.00	12.00	0	0	10.00	1.00	0	0	0
1253.00	883.00	2.00	46.00	42.00	58.00	5067.00	0	0	0	13.00	0
911.00	379.00	64.00	13.00	14730.00	724.00	-0	-0	2.00	653.00	794.00	313.00
960.00	0	-0	59.00	-0	2.00	-0	1.00	-0	0	-0	5.00
1265.00	158.00	4.00	17.00	0	119.00	16651.00	0	0	2.00	4.00	0
1279.00	0	834.00	5.00	0	1240.00	6995.00	0	214.00	0	214.00	0
1295.00	5977.00	196.00	0	2.00	442.00	513.00	0	948.00	2.00	0	0
1312.00	260.00	156.00	0	0	175.00	1038.00	0	2985.00	0	0	0
1325.00	58.00	0	0	0	521.00	438.00	0	4292.00	0	0	0
1335.00	8.00	0	0	0	508.00	38.00	0	413.00	0	0	0

CLADOCERA

Date	Bosm	Daph	Cerio	Alona	Chyd
189.00	9477.00	141.00	0	0	0
203.00	809.00	156.00	0	0	0
210.00	298.00	0	0	0	0
218.00	99.00	14.00	0	0	0
231.00	56.00	0	0	0	0
257.00	85.00	0	0	0	0
272.00	352.00	68.00	0	0	0
286.00	1035.00	114.00	47.00	0	0
312.00	2071.00	9.00	0	19.00	0
340.00	379.00	0	0	0	0
372.00	180.00	9.00	0	0	0
404.00	28.00	0	0	0	0
446.00	0	0	0	0	0
460.00	9.00	7.00	0	0	0
500.00	14.00	1.00	0	0	0
522.00	2081.00	11.00	0	0	0
533.00	173.00	20.00	0	0	2.00
540.00	391.00	46.00	0	0	0
547.00	166.00	7.00	10.00	0	7.00
554.00	1111.00	30.00	0	0	0
561.00	797.00	0	0	0	0
568.00	383.00	0	0	0	0
578.00	76.00	7.00	0	0	0
583.00	40.00	0	0	0	0
589.00	1.00	0	0	0	0
596.00	15.00	0	0	0	0
603.00	15.00	0	0	0	0
617.00	1.00	4.00	0	0	0
620.00	0	0	0	0	83.00
636.00	4.00	2.00	0	0	16.00
662.00	0	53.00	0	0	1.00
703.00	3.00	296.00	0	1.00	22.00
740.00	11.00	346.00	0	5.00	6.00
777.00	0	329.00	0	0	1.00
802.00	2.00	396.00	0	1.00	0
823.00	3.00	615.00	0	0	4.00
836.00	2.00	238.00	0	0	0
863.00	0	163.00	184.00	1.00	13.00
870.00	33.00	557.00	0	21.00	297.00
877.00	32.00	524.00	1.00	19.00	410.00
884.00	11.00	157.00	0	2.00	0
905.00	12.00	22.00	2.00	12.00	1162.00
933.00	108.00	212.00	0	79.00	12.00
954.00	24.00	199.00	0	26.00	4.00
940.00	76.00	162.00	1.00	13.00	7.00
968.00	12.00	19.00	99.00	0	4.00
889.00	550.00	25.00	0	0	2.00
897.00	185.00	42.00	1.00	0	46.00
919.00	50.00	27.00	10.00	0	63.00
925.00	681.00	104.00	29.00	0	50.00

CLADOCERA (cont'd)

Date	Bosm	Daph	Cerio	Alona	Chyd
946.00	69.00	14.00	12.00	2.00	61.00
974.00	432.00	67.00	61.00	0	170.00
998.00	1904.00	67.00	96.00	0	225.00
1007.00	1683.00	92.00	16.00	2.00	18.00
1023.00	1762.00	89.00	4.00	0	1.00
1202.00	0	0	0	0	0
1173.00	0	0	0	0	0
1142.00	0	0	0	0	0
1107.00	340.00	109.00	0	0	5.00
1073.00	364.00	45.00	0	0	21.00
1051.00	791.00	53.00	1.00	0	1.00
1235.00	106.00	18.00	1.00	0	2.00
1258.00	15.00	0	0	0	2.00
911.00	0	229.00	15.00	2.00	3.00
960.00	-0	316.00	349.00	87.00	-0
1265.00	4.00	0	0	0	0
1279.00	0	0	0	0	0
1295.00	0	0	0	0	0
1312.00	0	0	0	0	0
1325.00	0	0	0	0	0
1335.00	0	0	0	0	0